

University of Alberta

**Biophysical regulation of wildfire in the mixedwood boreal forest in the
context of climate change and forest management**

by

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A thesis submitted to the Faculty of Graduate Studies and Research in partial
fulfillment of the requirements for the degree of Doctor of Philosophy

in

Conservation Biology

Department of Renewable Resources

Edmonton, Alberta

Fall, 2007



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Your file Votre référence
ISBN: 978-0-494-33002-9
Our file Notre référence
ISBN: 978-0-494-33002-9

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Abstract

The dominant natural disturbance in the mixedwood boreal forests of Alberta is wildfire caused by lightning. The relative importance and interactive effect of variability in biotic (vegetation) and abiotic (weather) factors in regulating wildfire is controversial, as is the extent that recent changes in climate and/or land cover have contributed to fire activity and will continue to contribute in the future.

I use empirical data to demonstrate that patterns of lightning fire initiation are controlled by variability both in weather and forest composition that exist across landscapes in the mixedwood boreal forest. Fire initiation is reduced in landscapes with increasing amounts of aspen (*Populus tremuloides* Michx.) but increases with area of white spruce (*Picea glauca* (Moench) Voss) while accounting for variation in weather and lightning conditions. These two tree species form a principal successional trajectory in the region such that the likelihood of initiation changes through space and time from this biotic, endogenous regulation.

I demonstrate that forest management influences the fire regime we currently observe and will continue to do so in the future. Initiation increases in recently harvested landscapes and decreases in recently burned landscapes; harvesting does not emulate fire from the perspective of fire itself. I use simulation experiment to show this dissimilarity could manifest as an accelerated, compound disturbance inconsistent with the natural disturbance regime.

Climate change is projected to increase fire activity in western Canada. I use fire weather data from the Canadian Regional Climate Model with parameters estimated from observed, historical data to predict the effect of changing climate on lightning fire activity and contrast these to the outcome of simulation experiments where changes in fire weather and landscape composition (*via* fire, management and stand dynamics) occur simultaneously and interactively. Simulations showed forest management increases area of young, deciduous forest that reduces long-term area burned. This reduction is attractive for climate change mitigation however forest structure in harvested landscapes differs substantially from un-harvested landscapes. I examine an alternative regeneration policy to minimize these differences. Climate change increases area burned, but not enough regenerates as deciduous forest to cause long-term self-regulating feedback to fire.

Acknowledgements

I am very grateful to Steve Cumming, a wonderful supervisor who let me march to the beat of my own drum, but batted me with constructive criticism and discussion when required. Special thanks to Mike Flannigan and Fiona Schmiegelow for continued support during my dissertation work and to committee members Subhash Lele and (initially) Ross Wein.

Many people have contributed to this dissertation with their support and friendship, including: Kat Benedict, Lea Storry, Brian Starzomski, Eva Thompson, Sheila Potter; specific thanks to all those in the Schmiegelow Lab and especially Sophie Czetwertynski, Nadele Flynn and Keana, Kathryn Martell and Kaia, Brian Love, Layla Neufeld, Janie Wolken, Jared LeBoldus, Miles Dyck, Mark Hebblewhite and Keta, Kristin Campbell, John Gotwals, Krista Frolich and Finnegan, Chris Fisher, Marc Parisien (CFS), the late Bernie Todd (CFS), and Cordy Tymstra (ASRD). Ron Brooks and Phil Taylor have been an important influence as supervisors through my B.Sc. and M.Sc. where they fostered independent and creative thinking.

I am extremely grateful to my family (Mom, Dad and Carolyn) for always having an interest in my work and supporting my (however dubious) decisions. Cheers to Rus, the bestestest dog this girl could ever have.

Funding for this research was provided by NSERC, the Canadian Forest Service, the University of Alberta, the Department of Renewable Resources, ESRI, the Prairie Adaptation Research Collaborative, Canadian Forest Products Limited, John and Patricia Schlosser, and the Canadian Interagency Forest Fire Centre.

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Chapter 1. Introduction

Disturbance events punctuate the dynamics of ecosystems (Pickett and White 1985). These events are both reactive and creative: disturbances occur as a function of biophysical conditions (Schoennagel et al. 2004), organisms adapt to facilitate survival and/or reproduction in a certain succession of disturbance environments (Dobzhansky 1956) such that the diversity of communities is affected by rates of disturbance (Pickett and White 1985), and ecological systems can re-organize after disturbance as a function of novel environmental opportunities (Gunderson and Holling 2002). Disturbance events occur at any scale; Pickett et al. (1989) define a disturbance as 'change in the minimal structure of an ecological object caused by a biotic or abiotic factor external to the level of interest'. For example, if a stand of jack pine (*Pinus banksiana* Lamb) forest is the ecological object of interest, a fire that burns this stand is a disturbance. If the cone of a jack pine is the object of interest, then its consumption by a white-winged crossbill (*Loxia leucoptera* J. F. Gmelin) is a disturbance. Holling (1973) defines resilience as the persistence of relationships within a system, and is a measure of the ability of a system to absorb changes of state variables, driving variables, and parameters, and still persist. Key components of this definition are the changes caused by and to disturbances, and the effect of these changes on the persistence of a system in a given state. Understanding the causes and effects of disturbances contributes to understanding nature's variability.

The focus of my dissertation is the ecological disturbance to mixedwood boreal forest landscape structure in Alberta, western Canada, caused by lightning fires. I assess the biophysical factors regulating landscape patterns of lightning fire initiation, and the effects that projected changes in these biophysical factors could have on disturbance regime and the state of the system. Fire is the dominant natural disturbance in the mixedwood boreal section of the boreal plains ecozone (Figure 1-1; Stocks et al. (2002)). The outcome of accumulated fire events within a given time and space is used to describe the fire regime (pattern), including: type, intensity, severity, frequency, timing (season), and size (Gill 1975). The

behaviour (process) within each fire event is regulated by the interaction of weather, vegetation and, to some degree, topography (Cary et al. 2006) that control combustion based on the moisture (Van Wagner 1987) and structure of fuels (Rothermel 1972). The relative importance and interactive effect of weather and vegetation is controversial and has been strongly debated in the past (Bessie and Johnson 1995; Johnson et al. 1998; Hély et al. 2001; Schoennagel et al. 2004; Westerling et al. 2006; Cary et al. 2006; Krawchuk et al. 2006) with suggestion that weather conditions dominate patterns of fire activity (Bessie and Johnson 1995; Cary et al. 2006). Recent evidence suggests that the relative importance of these biotic and abiotic factors varies (Figure 1-2; Johnson et al. 1998; Schoennagel et al. 2004; Krawchuk et al. 2006), based on the heterogeneity of driving factors at the scale of study.

Ongoing human development and global warming are expected to generate unprecedented changes to global systems over the next century (Millenium Ecosystem Assessment 2005; IPCC 2007). Northern ecosystems are particularly sensitive to changes in climate and development (Bonan and Shugart 1989; Weber and Flannigan 1997; Chapin et al. 2000; Smith and Lee 2000) generating concern that boreal forests will change substantially in the near future (Chapin et al. 2000). Fire is predicted to be a key process involved with system changes due to tight link between weather, vegetation and fire (Weber and Flannigan 1997). For example, infrastructure development and forest harvesting (Schneider et al. 2003) will result in direct changes to landscape structure (composition, configuration and connectivity) that may alter fire activity (Quintilio et al. 1991; Hirsch et al. 2001). Changes in climate will also contribute directly to changes in structure as growing conditions for common tree species such as aspen (*Populus tremuloides* Michx) and white spruce (*Picea glauca* (Moench) Voss) are altered (Hogg and Bernier 2005). Changes in fuel structure may interact with the directional change in climate via more extreme fire weather generating more fire (Flannigan et al. 2005) which could also affect the effect of fire suppression initiatives (Fried et al. 2004).

The goal of science is to acquire knowledge about the natural world through empirical evidence. Here, my goal is to acquire knowledge about the biophysical regulation of wildfire in the mixedwood boreal ecosystem in the context of regional climate change and forest management through the analysis of empirical data and simulation experiment at landscape and regional scales (Figure 1-2). This research includes the process and patterns of wildfire, but has considerable focus on lightning fire initiation. Lightning currently generates the majority of fires in the mixedwood boreal forest but humans also initiate fires (Arienti et al. 2006) and have actively manipulated wildfire in the past (Murphy 1985; Bond and Keeley 2005) so that our current environment reflects millennia of interactions between humans and their environment (Willis and Birks 2006). The influence of climate change and forest management on the mixedwood boreal forest has been explored at various spatial and temporal scales (Johnston et al. 2001) however regional initiatives focused at understanding the independent and interacting contribution of variability in biotic (vegetation) and abiotic (weather) factors to wildfire activity are lacking, or isolated to thought experiments. When I began this research I had a broad hypothesis that changes in climate would have a relatively slower or smaller effect on lightning fire initiation (and thus fire behaviour and fire regime) in comparison to ongoing and faster changes in forest composition generated by forest management within Alberta's mixedwood boreal forests (Figure 1-1). As with any project, I veered from and returned to this course, and fleshed out a few edges. The proceeding four chapters present this research.

Fires generate heterogeneous landscape patterns that are readily observed and can be used to quantify the recent historical fire regime. Spatial and temporal records provide the location and time of recent fire events, weather, and land cover in Alberta. In Chapter 2, I used a logistic regression model to quantify the independent, relative influence of variation in landscape composition and fire weather conditions on patterns of lightning fire initiation in the mixedwood boreal forest. I used data from a sampling period from 1983 to 1993 when wildfire detection was relatively consistent, and large-scale harvesting in the study region

had not yet begun. This work was published in *Ecology* as Krawchuk et al. (2006).

In Chapter 3, I used a zero-inflated negative binomial regression model to re-visit the influence of biophysical variables on patterns of lightning fire initiation. I used data from a sampling period from 1994 to 2001 when large-scale forest management activity had altered the vegetation composition of the region through the added contrast of harvested and regenerating stands in addition to those initiated by wildfire. I integrated parameter estimates from the regression model into an existing stochastic regional simulation model of boreal mixedwood forest dynamics, Tardis (Cumming and Armstrong 2004) to study the long-term local and regional effects of these relationships. This work has been submitted to *Ecological Applications*.

In Chapter 4, I used a zero-inflated negative binomial regression model to quantify the independent influence of fire weather variables on lightning fire initiation (i.e., in the absence of vegetation effects), and used parameter estimates from this model along with weather data generated by the Canadian Regional Climate Model (Laprise et al. 1998) to predict future changes in initiation. I integrated these predictions with existing empirically-based models of fire growth (Cumming 2005) and final size (Cumming 2001) for the region to generate estimates of future area burned expected from changes in the rate of lightning fire initiation. This work has been submitted to the journal *Climatic Change*.

In Chapter 5, I used a simulation experiment to quantify the effect of independent and interacting influences of forest management and climate change on emergent patterns of wildfire activity (initiation and area burned) by developing a novel fire-climate module within the existing simulation model, Tardis (Cumming and Armstrong 2004). I used a suite of metrics to test three hypotheses formulated around the concepts of 1) a negative feedback in fire activity caused by a near-term increase in area burned initiated by climate change causing a subsequent, relative decrease in long-term fire activity due to a reduction in flammable forest fuels, 2) the effect of forest management on current and future fire activity via the alteration of landscape composition, and 3) the

effect of an alternative forest management policy aimed to reduce fire initiation via changes in forest regeneration rules.

The complex feedbacks and interactions within ecosystems add layers of uncertainty to any projections of boreal system state or function in the future. Given the intricacies of this system and our incomplete knowledge of any component, it is helpful in moving forward to remember Box's (1979) statement that "while all models are wrong, some are useful".



Figure 1-1. The mixedwood forest section (Rowe 1972) makes up the majority of the boreal plains ecozone (Wiken 1986), illustrated in grey hatching. The shaded square illustrates the size and location of the study area. The Province of Alberta, Canada contains most of the mixedwood boreal forest.

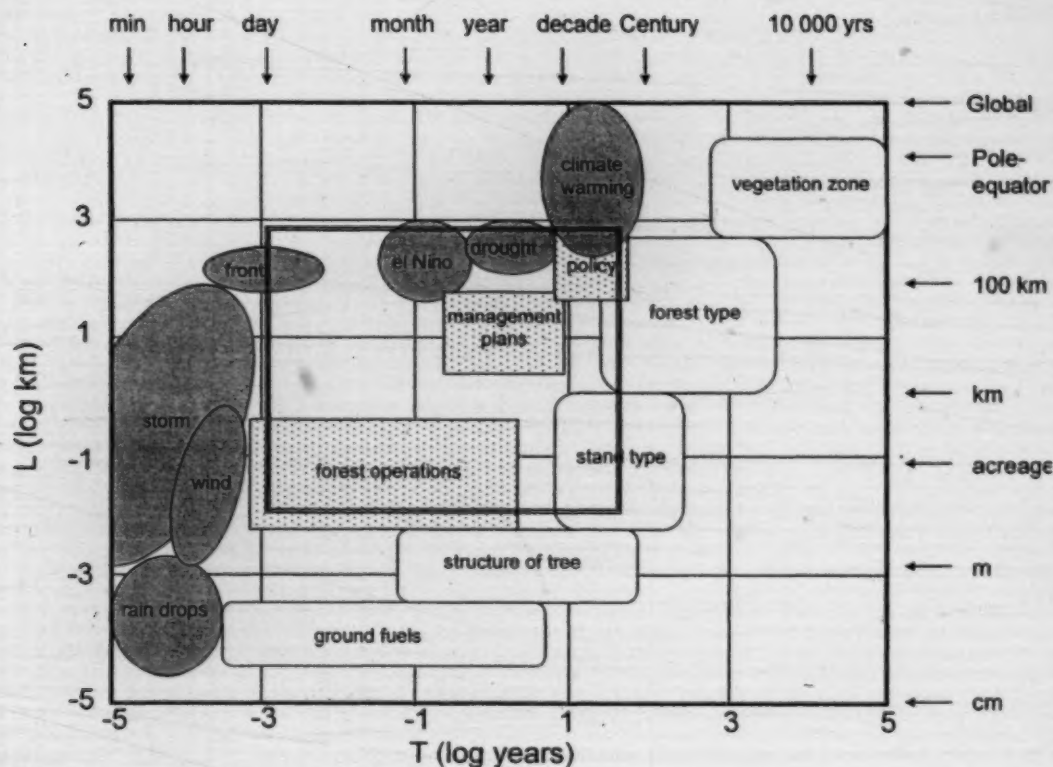


Figure 1-2. Biotic and abiotic factors affect wildfire at many spatial and temporal scales. Forest management interacts within this hierarchy. Factors contributing to variation in weather and moisture (abiotic) are illustrated in grey while factors contributing to variation in vegetation and fuel composition (biotic) are illustrated in white. The scales of forest management activities are illustrated in stippled boxes. The scale of focus for the thesis is outlined by a double-lined box.

Literature Citations

- Arienti, M.C., S.G. Cumming and S. Boutin. 2006. Empirical models of forest fire initial attack success probabilities: the effects of fuels, anthropogenic linear features, fire weather, and management. *Canadian Journal of Forest Research* 36:3155-3166.
- Bessie, W.C. and E.A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forest. *Ecology* 76:747-762.
- Bonan, G. B. and H.H. Shugart. 1989. Environmental factors and ecological processes in boreal forests. *Annual Review of Ecology and Systematics* 20:1-28.
- Bond, W.J. and J.E. Keeley. 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution* 20:387-394.
- Box, G.E.P. 1979. Robustness in the strategy of scientific model building. Pages 201-236, in R.L. Launer and G.N. Wilkinson, editors. *Robustness in statistics*. Academic Press, New York.
- Cary, G.J., R.E. Keane, R.H. Gardner, S. Lavorel, M.D. Flannigan, I.D. Davies, C. Li, J.M. Lenihan, T.S. Rupp and F. Mouillot. 2006. Comparison of the sensitivity of landscape-fire-succession models to variation in terrain, fuel pattern, climate and weather. *Landscape Ecology* 21:121-137.
- Chapin, F.S., A.D. McGuire, J. Randerson, R. Pielke, D. Baldocchi, S.E. Hobbie, N. Roulet, W. Eugster, E. Kasischke, E.B. Rastetter, S.A. Zimov and S.W. Running. 2000. Arctic and boreal ecosystems of western North America as components of the climate system. *Global Change Biology* 6:211-223. Notes: Supplement: Suppl. 1
- Cumming, S.G. 2001. A parametric model of the fire-size distribution. *Canadian Journal of Forest Research* 31:1297-1303.
- Cumming, S.G. 2005. Effective fire suppression in boreal forests. *Canadian Journal of Forest Research* 35:772-786.
- Cumming, S. G. and G.W. Armstrong. 2004. Divided landbase, overlapping tenures, and fire risk. *Forestry Chronicle* 80:478-484.
- Dobzhansky, T. 1956. What is an adaptive trait? *American Naturalist* 90:337-347.
- Fried, J.S., M.S. Torn and E. Mills. 2004. The impact of climate change on wildfire severity: A regional forecast for northern California. *Climatic Change* 64:169-191.

- Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner and B.J. Stocks. 2005. Future Area Burned in Canada. *Climatic Change* 72:1-16.
- Gill, A. M. 1975. Fire and the Australian flora: a review. *Australian Forestry* 38:4-25.
- Gunderson, L.H. and C.S. Holling. 2002. *Panarchy*. Island Press.
- Hély, C., M.D. Flannigan, Y. Bergeron, and D. McRae. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. *Canadian Journal of Forestry Research* 31:430-441.
- Hirsch, K., V. Kafka and Todd, B. 2001. Using forest management techniques to alter forest fuels and reduce wildfire size: and exploratory analysis. *in* R.T. Engstrom and W.J. de Groot, *Proceedings of the 22nd Tall Timbers Fire Ecology Conference*. Fire in temperate, boreal, and montane ecosystems.
- Hogg, E.H.T. and P.Y. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. *Forestry Chronicle* 81: 675-682.
- Holling, C.S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4:1-23.
- IPCC. 2007. *Climate Change 2007: The physical science basis. Summary for policy makers. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*.
- Johnson, E.A., K. Miyanishi and J.M.H. Weir. 1998. Wildfires in the western Canadian boreal forest: Landscape patterns and ecosystem management. *Journal of Vegetation Science* 9:603-610.
- Johnston, M., E. Wheaton, S. Kulshreshtha, S. V. Wittrock and J. Thorpe. 2001. *Forest ecosystem vulnerability to climate: An assessment of the western Canadian boreal forest*. Saskatoon, Saskatchewan, Canada., Saskatchewan Research Council.
- Krawchuk, M.A., S.G. Cumming, M.D. Flannigan and R.W. Wein. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. *Ecology* 87:458-468.
- Millenium Ecosystem Assessment. 2005. *Ecosystems and human well-being: Synthesis*. Island Press, Washington.
- Murphy, P.J. 1985. *History of forest and prairie fire control policy in Alberta*. Report T/77, Alberta Energy and Natural Resources, Edmonton, Alberta.
- Pickett, S.T.A. and P.S. White. 1985. *The ecology of natural disturbance and*

- patch dynamics. Academic Press, Orlando.
- Pickett, S.T.A., J. Kolasa, J.J. Armesto and S.L. Collins. 1989. The ecological concept of disturbance and its expression at various hierarchical levels. *Oikos* **54**:129-136.
- Quintilio, D., M.E. Alexander and R.L. Ponto. 1991. Spring fires in a semimature trembling aspen stand in central Alberta. Information Report NOR-X-323. Forestry Canada, Northwest Region.
- Rothermel, R.C. 1972. A mathematical model for predicting fire spread in wild land fuels. USDA Forest Service, Research paper INT-115.
- Rowe, J.S. 1972. Forest regions of Canada. Publication Number 1300. Canadian Forest Service.
- Schneider, R.R., J.B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the Western Canadian Sedimentary Basin: A modelling approach. *Ecology and Society* **7**(1):8.[online].
URL:<http://www.ecologyandsociety.org/vol7/iss1/art8>
- Schoennagel, T., T.T. Veblen and W.H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. *Bioscience* **54**:661-676.
- Smith, W. and P. Lee. 2000. Global Forest Watch Canada. Canada's forests at a crossroads: an assessment in the year 2000. Washington, D.C., World Resources Institute.
- Stocks, B.J., J.A. Mason, J.B. Todd, E.M. Bosch, B.M. Wotton, B.D. Amiro, M.D. Flannigan, K.G. Hirsch, K.A. Logan, D.L. Martell and W.R. Skinner. 2002. Large forest fires in Canada, 1959-1997. *Journal of Geophysical Research-Atmospheres* **108**.
- Van Wagner, C.E. 1987. Development and structure of the Canadian Forest Fire Weather Index System. Forestry Technical Report 35. Canadian Forestry Service. Ottawa, Canada.
- Weber, M.G. and M.D. Flannigan. 1997. Canadian boreal forest ecosystem structure and function in a changing climate: Impact on fire regimes. *Environmental Reviews* **5**:145-166.
- Westerling, A.L., H.G. Hidalgo, D.R. Cayan and T.W. Swetnam. 2006. Warming and earlier spring increase western US forest wildfire activity. *Science* **313**: 940-943.
- Wiken, E.B. 1986. Terrestrial Ecozones of Canada. Ecological Land Classification, Series No. 19. Environment Canada. Hull, Quebec. 26pp. + map.

Willis, K. J. and H.J.B. Birks. 2006. What is natural? The need for a long-term perspective in biodiversity conservation. *Science* 314:1261-1265.

Chapter 2. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest*

Introduction

Natural disturbances are characteristic of all ecosystems, stimulating a cycle of release, reorganization, and regeneration (Gunderson and Holling 2002). Biotic and abiotic processes regulate disturbance events (e.g., Paine and Levin 1981) and the dynamics of an ecosystem depend on the comparative endogeny or exogeny of these processes (Didham and Watts 2005). Lightning fires are the dominant natural disturbance in much of the North American boreal forest (Stocks et al. 2003). The relative importance of biotic and abiotic factors regulating these fires is controversial (Bessie and Johnson 1995; Hély et al. 2001; Schoennagel et al. 2004). Our study considers one aspect of lightning fire behaviour – initiation. Here, initiation is the outcome of a sequence of physical events and a detection system, resulting in the record of a fire caused by lightning (Cunningham and Martell 1976).

An initiation can be described mechanistically as the interaction between atmospheric and terrestrial conditions. Broad scale weather patterns determine the location of lightning events that will provide the energy for ignition to occur. The efficiency of individual lightning strikes in igniting a fire is affected by variation in: lightning properties such as polarity, multiplicity, or current (Fuquay et al. 1979); moisture properties of forest fuel resulting from recent weather conditions including precipitation, temperature, wind and humidity (Van Wagner 1987); or rates of combustion that vary with fuel type (Rothermel 1972). Lab studies have shown that moisture content and fuel depth are important elements of ignition (Latham and Schlieter 1989). Fuel type can influence initiation through variation in cellular structure of the fibre and moisture content as well as the size and shape of fuel on the forest floor (Anderson et al. 2000). Survival from smoldering to an active fire is dependent on organic and soil properties (Hartford 1989). Under appropriate joint conditions a strike may ignite fuels, continue to combust, and

* A version of this chapter has been published. Krawchuk et al. 2006. *Ecology* 87:458–468.

persist to become a fire large enough to be detected – an initiation. Given a large sample, we view initiation as a random spatio/temporal point process with parameters conditional on covariates for each limiting factor. These include: lightning strike data to determine where initiation potential existed, weather data to assess fuel moisture conditions, and land-cover data to assess the fuel types existing at the time of initiation.

Many previous studies (e.g., Fowler and Asleson 1984; Flannigan and Wotton 1991; Renkin and Despain 1992; Nash and Johnson 1996; Diaz-Avalos et al. 2001; Wierzchowski et al. 2002; Podur et al. 2003) have quantified the initiation process using some of the relevant covariates measured at various spatial and temporal scales. However, none incorporated the full complement of features critical to identifying independent correlates of initiation. The objective of our study was to quantify the relative contribution of weather and fuel composition to the initiation process in the western mixedwood boreal forest ecosystem. We considered weather as an abiotic, exogenous factor and fuel composition as a biotic, endogenous factor contributing to a disturbance event. By advancing our understanding of the environmental patterns influencing the distribution of initiations we contribute insight to the processes regulating natural dynamics in the boreal forest and to scenarios that could arise from the interaction of unprecedented changes in climate (Flannigan et al. 2001) and land cover (Schneider et al. 2003) anticipated for the region in the near future.

Methods

Study region. Our study region was 91 000 km² of mixedwood boreal forest (Rowe 1972) in central-eastern Alberta, Canada (Figure 2-1). The regional climate is continental with mean winter and summer temperatures of -13.2°C and +13.5°C, respectively and total precipitation in summer and winter in the range of 240 mm and 64 mm, respectively (Strong and Leggat 1992). Elevation varies gradually from approximately 200 to 1000 m. Glacial movement through the region has resulted in rolling morainic deposits on the uplands and smoother glacio-lacustrine deposits on the lowlands (Rowe 1972). The mixedwood is the

most productive and diverse of boreal ecosystems in North America (see Chen and Popadiouk 2002 for a review). Forest tree communities (Table 2-1) are stratified by soil moisture regime (Strong and Leggat 1992). Upland mesic sites are dominated by mixtures of trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.) or white spruce (*Picea glauca* (Moench) Voss). Community development at these sites depends on variation in disturbance history, composition, and spatial context (e.g., stochastic availability of propagules from neighbouring stands). Paper birch (*Betula papyrifera* Marsh) is present but pure stands are rare. White spruce and balsam fir (*Abies balsamea* (L.) Mill) dominate older stands but pure stands are not currently abundant. Xeric sites are vegetated by jack pine (*Pinus banksiana* Lamb), sometimes in association with aspen. Moist, poorly drained hygric sites are vegetated by black spruce (*Picea mariana* (Mill.) B.S.P.). There are extensive wetlands including open areas of bog, fens, swamps and marshes. Tamarack (*Larix laricina* (Du Roi) Koch) is the dominant tree species in wetlands.

Data and exploratory analysis. The study region was divided into three-dimensional space-time samples, called voxels, with the units landscape and year. The Province of Alberta is partitioned using an administrative grid system of townships each measuring approximately 9 600 ha and varying slightly due to geographic corrections. Each township in the study region was considered as the spatial sampling unit, hereafter referred to as a landscape (Figure 2-1). Data were collected for each landscape between 1983 and 1993 and summarized for each year providing the annual, temporal dimension to the voxel.

We described the relationship between the occurrence ([1] below) of an initiation in a voxel with respect to a suite of biotic and abiotic variables [2-7] described by indices of: 2) fire weather (weather reflecting fuel moisture relevant to fire initiation and behaviour), 3) climate, 4) lightning, 5) joint fire weather-lightning, 6) geography and 7) forest composition using logistic regression. A summary of variables is available in Table 2-1. Daily meteorological data were collected during the fire season and summarized to describe weather-related variation per voxel. The exception was the climate index where daily

meteorological data were summarized over the entire region for each year. Geography was a static description of landscape location and did not vary through time. Forest composition was described for each voxel and varied through time as a result of natural and anthropogenic disturbance. The eleven years of data represented a time series prior to a marked, intensified forest harvesting in the study region (Schneider et al. 2003).

1) Lightning fire initiation. Alberta Sustainable Resources Development (ASRD) provided data describing the location and date of initiations. Un-detected or un-recorded fires may have occurred in the region but fire detection efficiency appears to have been constant over the study interval (Cumming 2005). A binary response variable was selected to describe the presence or absence of an initiation in each voxel, such that $y_{ij} = 1$ if at least one initiation occurred in landscape i in year j . Multiple initiations were infrequently detected but these events were omitted to preclude the statistical dependence of events resulting from the same weather/forest conditions.

The $\sum y_{ij} = 1280$ over the study period, and $p_{ij} = 0.12$. The mean annual number of initiations in the region was 116 (s.d. = 59). We expected spatial and/or temporal structure might exist in the processes generating all variables (Legendre et al. 2004) however Bonferroni-corrected Mantel r -statistics (Mantel 1967; Legendre and Legendre 1998) indicated there was no significant autocorrelation of y_i within each year j .

2) Fire weather. We summarized daily weather and fuel moisture conditions relevant to fire ignition provided by The Canadian Forest Fire Weather Index System (FWI system; Van Wagner 1987) to produce three fire weather indices for each voxel. We selected the Fine Fuel Moisture Code (FFMC), Duff Moisture Code (DMC) and Initial Spread Index (ISI). FFMC and DMC represent moisture content of litter or fast-drying fine fuels and loosely compacted organic layers of moderate depth, respectively. ISI includes wind speed in calculation and is theoretically relevant to the initial spread rate of a fire independent of fuel effects (Van Wagner 1987). These codes were calculated from daily measurements of temperature, relative humidity, wind speed and precipitation

taken from an array of 65 provincial weather stations operated and calibrated by ASRD in and around the region and interpolated across the region. We estimated a critical value where fuel moisture was 'receptive' to fire propagation by assessing the cumulative distribution of the recorded daily FFMFC, DMC and ISI codes for each initiation in the data set. The critical value was selected as the code above which 85 % of initiations were detected. We tallied the sum number of days per voxel when the code exceeded the critical value to create three fire weather indices. These indices, FFMFC_{DAYS}, DMC_{DAYS}, and ISI_{DAYS} (Table 2-1) were used to describe variation in receptive fire weather and fuel moisture.

The three daily codes were interpolated at a resolution of 1 km across the study region on each day of the fire season over 11 years ($n = 3 \text{ codes} \times 155 \text{ days} \times 11 \text{ years} = 5115$) by inverse-distance weighting (Flannigan and Wotton 1989) in ARC-GIS (ESRI 2002) using the six nearest weather stations. The interpolation of each code was cross-validated at 20 stations that covered a range in elevation. Data from randomly selected days were removed, and the codes were interpolated for these weather stations. For each code we used linear regression to test if the slope of the relationship between interpolated and measured values was significantly different from unity ($\alpha = 0.05$). To test the sensitivity of the generic interpolation to known adiabatic lapse rates we used regression residuals from above to test if the slope of their relationship with elevation was different from zero ($\alpha = 0.05$). Interpolations underestimated ISI but the relationship was linear. Otherwise, all tests indicated that the inverse-distance-weighting technique was adequate (Table 2-2).

Threshold values of receptive fire weather calculated for FFMFC, DMC and ISI were 87, 34 and four. We lowered the critical ISI from four to three as a simple adjustment for the bias in interpolation described above. These values agree closely with analyses from a similar forest region in Saskatchewan, Canada (Anderson and Englefield 2001).

3) Climate. A synopsis of annual variation in regional atmospheric conditions – e.g., a “dry year” – was quantified using the seasonal severity rating (SSR) from the FWI system (Van Wagner 1987). SSR was calculated as the mean

annual Daily Severity Rating of all weather stations in the region. SSR varied from 0.8 to 1.4 (mean = 1.06, s.d. = 0.17). A generalized linear regression model using a Poisson family suggested a log-linear relationship between counts of annual initiations and SSR ($\beta = 2.12$, s.e. = 0.13, $n=11$).

4) Lightning indices. We created two indices to describe the spatio-temporal distribution of lightning activity among voxels (Table 2-1). Lightning data were obtained from ASRD detection stations across the province (Figure 2-1): The location of each lightning flash is estimated using time difference and triangulation techniques along with estimates of the polarity, multiplicity of strokes (number of return strokes in a flash), time, and peak current. The relationship between lightning characteristics and ignition probability is incompletely understood and not all strikes have characteristics required to initiate a fire (Rakov and Huffines 2003). A long continuous current can not be detected with current technology and not all lightning strikes are recorded. As such, we considered all strikes equally in this study. The first index was the summed number of days in a voxel where at least one lightning flash was detected (L_{DAYS}), the second index was the cumulative number of detected lightning strikes (L_{COUNT}) in a voxel.

Nimchuk (1989) estimated spatial error of Alberta's lightning detection system at 4 – 14 km. This error is larger than the spatial resolution of each landscape. We assumed that if detected lightning events were autocorrelated across multiple landscapes and the direction of error was unbiased, the overall effect of spatial error and misclassification at the voxel scale is dampened. Variogram analysis (Cressie 1991) illustrated that daily lightning activity was spatially structured within years across the study region suggesting that the influence of positional inaccuracy reported for strikes could be buffered by the spatial extent of storm activity and the resolution of our study voxels. Both indices varied weakly with elevation (Figure 2-2). There was no evidence to suggest the location of lightning detectors (Figure 2-1) influenced lightning indices.

5) Joint weather-lightning indices (JWL). We were motivated to create the joint fire weather-lightning indices from studies documenting an inverse relationship between lightning efficiency (number of fires/number of strikes) and fuel moisture (Nash and Johnson 1996) suggesting the co-occurrence of 'appropriate' daily fire weather conditions and daily lightning should better explain the distribution of initiations than the independent distributions each. Three joint weather-lightning indices (Table 2-1: $FFMC_{JWL}$, DMC_{JWL} , ISI_{JWL}) were calculated for each voxel to describe the sum number of days when fuel moisture conditions were receptive to ignition and lightning was detected. This index joins the 'fire weather' indices with the L_{DAYS} 'lightning index'. Previous studies (Kourtz and Todd 1991) have suggested that large storms produce more lightning but also more precipitation. We calculated weighted JWL indices ($wFFMC_{JWL}$, $wDMC_{JWL}$) to account for this convex relationship between daily lightning density and ignition potential. For each day during the fire season landscapes with no lightning and/or sub-threshold fire weather were classified as zero, between one and five strikes of lightning and above-threshold fire weather as one, and five or more strikes and above-threshold fire weather with a value of 0.2. Values for the lightning cut-points were based on Kourtz and Todd (1991).

6) Geography. We described the elevation (ELEV) of each landscape at its centroid using a Digital Elevation Model (DEM). The DEM was a 100 m resolution grid created from digitized 1:20 000 topographic base-maps with 10 m contours provided by Alberta-Pacific Forest Industries Inc. The NAD 27 Universal Trans Mercator measure of northing (NORTH) was included in analyses as a spatial proxy for unmeasured gradients. Northing ranged from a UTM of 6 100 565 to 6 428 588.

7) Forest composition. We used the Alberta Phase 3 Forest Inventory (Anonymous 1985) circa 2000 to describe forest composition within each voxel as a broad scale indicator of fuel availability and variability. The inventory was carried out between 1970 and 1984 based on the interpretation of 1:50 000 leaf-on aerial photographs, supplemented by ground-truthing (Anonymous 1985). The inventory has been updated annually by the province for fire, harvesting and other

land conversion since its compilation. We simplified the inventory classification to 10 stand types based on dominant canopy species, composition and stand density. We used the seven most common stand types to describe the compositional area (ha) of each voxel (Table 2-1). These included low density aspen-deciduous (A_L), high density aspen-deciduous (A_H), white spruce (S_W), black spruce (S_B), jack pine (P_J), recently burned (B), and open wetland/low density scrub (O). Remaining fuel types accounted for < 2 % of the study region and were omitted from analyses. Landscape composition in years prior to fire and harvest was recovered from detailing in map annotations and notation used in the update process (as described in Cumming (2001)) enabling us to quantify changes in composition for each voxel resulting from disturbances. We assumed forest composition was otherwise constant over the study interval. Forest types varied across the region and provided the contrast required for this study (Figure 2-2). Of the 1056 landscapes within the region, 946 were selected for study where classifiable terrestrial surface area was greater than 5 000 ha. This criterion excluded landscapes with small area, large areas of water or human development. These 946 landscapes were sampled over 11 years providing 10406 voxels.

Logistic regression. We described the response variable, initiation per voxel, as a function of fire weather and lightning indices, forest composition and geography using logistic regression in a generalized linear model framework (McCullagh and Nelder 1989) using the R statistical language and environment (R Development Core Team 2005; version. 2.0.1). We verified the linearity of the relationship between initiation probability and all candidate covariates through the log odds. These relationships were linear with the exception of the JWL indices and the area of black spruce forest cover type. Log transformations of JWL indices induced linearity (0.5 was added to each index to accommodate transformation of zeros). We included a linear and quadratic term to account for non-linear relationship with black spruce composition. We found no relationship between the odds of initiation and landscape area (mean = 9140 ha, s.d. = 438 ha) so area was not considered in the remainder of this work.

We assessed the collinearity between and among covariate classes using Pearson's product-moment pair-wise correlation. The correlation among forest types had a maximum absolute value of $r = 0.36$ exhibited between open areas (O) and high density hardwood (A_H), remaining forest types were correlated within ± 0.3 indicating multi-collinearity in terms should not affect the analyses. Weather indices were inherently correlated since they included combinations of the lightning and weather data. The correlation among forest composition, fire weather and JWL indices among voxels did not exceed ± 0.1 indicating there was sufficient variability to assess the relative influence of forest composition and weather. Scatter plots illustrated low correlation among annual JWL indices, forest composition, geography and elevation (Figure 2-2). Our exploratory analyses indicated that there was no spatial autocorrelation of initiations among voxels suggesting that residuals should be spatially, independently distributed and that we had an effective sample size of 10406 (Legendre et al. 2004). As such, we began our analyses using a fixed effects logistic regression model and based selection of the most parsimonious model on Akaike's Information Criterion (AIC) using a backward elimination procedure (Burnham and Anderson 1998). A change in AIC among candidate models of greater than five was used to indicate a substantial change in the systematic descriptive ability of each model.

Our primary objective was to determine the independent influence of variation in weather and fuel on initiation. We used a hierarchical model selection approach to determine the best weather model (W) and best fuel model (F), then combined these to determine the best weather and fuel model (WF). First a global weather model (W_0) of all ten weather-related indices (Table 2-3) was reduced to the most parsimonious (best) weather model (W_1). Similarly, a global fuel model (F_0) of all seven forest composition variables was reduced to the best fuel model (F_1). W_1 and F_1 were invited to the global weather and fuel model (WF_0 ; Table 2-3) where two geographic measures (ELEV and NORTH) were also included. WF_0 was reduced to the best fixed effects model, WF_1 . We then assessed the independence of residuals from (WF_1) to determine if additional, un-observed processes might have generated spatial or temporal autocorrelation. There was

still no evidence of correlation in residuals among landscapes or as a function of year. Box-plots of residuals grouped by landscape suggested correlation within a landscape throughout the study period, but this was not structured by time or time interval. We compared WF_1 to a logistic mixed effects model (using Broström 2003) where a random effect with compound symmetry in the variance-covariance matrix was included to describe variation of the intercept for each landscape (WF_2). WF_2 assumes that stationary, yet unmeasured attributes influence the probability of initiation for voxels at a given landscape through time.

Our primary objective did not specify explicit hypotheses regarding interaction terms. We tested the hypothesis that the influence of fuel type would decrease in years with more extreme fire weather through interactions between forest composition terms and SSR using model WF_2 . It has been suggested that all forest types are equally flammable under extreme fire weather conditions (Renkin and Despain 1992; Bessie and Johnson 1995). For a given forest composition term, a difference in sign between the main effect and the interaction terms would provide evidence that the influence of fuel type was diminished in more extreme fire years.

We assessed the relative influence of forest composition and weather/moisture on initiation by comparing the variation explained by W_1 , F_1 and WF_1 using AIC scores and the Nagelkerke R^2 . Model fits were assessed with the le Cessie - van Houwelingen - Copas - Hosmer unweighted sum of squares test (Hosmer et al. 1997) and Receiver Operating Characteristic (ROC) plots. The area under the ROC curve measured the ability of a logistic regression model to predict the occurrence of an event within the initial data set where values above 0.5 indicate increasingly substantial explanatory power (Fielding and Bell 1997). We evaluated model residuals using spatial, temporal and unstructured diagnostics. We are not aware of any goodness of fit tests currently developed for logistic mixed effects models so we assume those calculated from the fixed effects models provide conservative estimates for WF_2 .

Results

Model WF_1 offered the lowest fixed effects AIC relative to the global model WF_0 . WF_1 and WF_2 retained four forest composition terms, two JWL indices, SSR, elevation and northing (Table 2-3). Inclusion of the quadratic term for black spruce (S_B^2) reduced the AIC by three and this was retained in WF_2 due to evidence of non-linearity. The maximum of this quadratic term occurred at a value of 3070 ha and 68 landscapes had black spruce in excess of this value. The change in AIC for SSR was large enough ($\Delta AIC = 35$) to maintain it was significant at a regional scale, with an effective sample size of 11 rather than 10406. A change in AIC of 16 between WF_1 and WF_2 suggested the inclusion of a random effect for landscape to the best fixed effects model explained sufficiently more variation in the data (Table 2-4). We proposed that spatio-temporal structure in JWL indices could lead to stable initiation potential within each landscape and be responsible for the residual autocorrelation being quantified by the random effects. We detected significant positive autocorrelation in the 11-year cumulative sum of each JWL index up to 100 km (10 landscapes) however we did not detect any corresponding spatial structure in the value of the random effects using variograms.

There was a significant, positive interaction between SSR and the S_w forest composition term when it was included in WF_2 ($\Delta AIC = 17$, $\beta = 11.48$, s.e. = 2.71, Figure 2-3). We conclude that the effect of fuel variation on initiation was amplified in more extreme fire years, rather than decreased. The SSR values for three (1985, 1990 and 1991) of eleven years were more extreme than the third inter-quartile range of a 20 year series (1973-1993) of data from the same region indicating these years could be considered extreme fire years in the region. There were no other significant interactions between forest composition types and SSR.

ROC area under the curve goodness-of-fit tests indicated WF_1 had substantial explanatory value (Table 2-3). The le Cessie - van Houwelingen - Copas - Hosmer unweighted sum of squares test for global goodness of fit also indicated that WF_1 was well-fit model with a p-value of 0.70. F_1 was a more parsimonious model than W_1 using AIC, although ROC-AUC values differed

only slightly (Table 2-3). R^2 values from the fixed effects models supported the relative superiority of F_1 over W_1 and the ΔAIC and effect size of parameter estimates from WF_2 (Table 2-3, Figure 2-4) demonstrated dominance in the magnitude of influence for forest composition relative to weather.

Figure 2-4 illustrates that initiation is influenced by both fuel and weather, but that fuel has the dominant effect. Expected probabilities were calculated using parameter estimates from WF_1 . A) illustrates a trade-off in the amount of aspen (A_H) and white spruce (S_W) in a landscape while maintaining 1000 ha of both burned (B) and black spruce (S_B) forest types and all other covariates are maintained at their mean value (Table 2-1); B) illustrates a similar trade between aspen (A_H) and black spruce (S_B).

Discussion

We demonstrated that forest composition and weather conditions independently influenced the annual probability of lightning fire initiation within a large area of the mixedwood boreal forest. The fuel effects strongly predominated, even under extreme annual weather conditions. The probability of initiation was greater in spruce (*Picea*) dominated landscapes than in deciduous, aspen (*Populus*) landscapes. This implies that initiation is more frequent in spruce and less frequent in deciduous stands. This effect was amplified for landscapes containing white spruce in years with more extreme fire weather, as measured by SSR.

The mixedwood boreal forest is a heterogeneous region characterized by its mixedwood stands that are typically composed of white spruce and trembling aspen. These stands develop through structural stages that usually include a change in species composition. Multiple developmental pathways exist for mixedwood stands in mesic areas (Chen and Popadiouk 2002). The state of a stand is expressed from the interaction of stand condition, context, and disturbance history (Chen and Popadiouk 2002; Albani et al. 2005). Trembling aspen is the typical regenerating species in a stand when it is present in the pre-fire landscape. This is a result of vegetative shoot development from root suckers.

White spruce may establish at stand initiation depending on seed source, micro-site conditions and inter-specific competition (Greene et al. 2004; Albani et al. 2005). White spruce is shade tolerant and will grow in the understory of aspen stands to be released to the overstory as aspen reaches maturity, dies and provides gaps in the canopy. Through this pathway, stands may burn, regenerate along a successional trajectory with early deciduous dominance and lower probabilities of initiation, then develop to spruce dominance and higher probabilities of initiation; a 'fire-prone' white spruce can be seen as growing up in a protective aspen nursery. We must consider that initiation patterns (and fire regimes) change over long time scales (e.g., multiple disturbances) and reflect the intrinsic stochasticity of mixedwood boreal stand development.

The variation of initiation among forest types can be interpreted through variation in quality and quantity of fuels among them. Hély et al. (2000) provide a thorough review of variation in fuel characteristics among stand types similar to those in our study. They suggest that stand fire hazard increased via the quality of surface fuels as a result of species replacement through time. Spruce stands tend to contain more flammable fuel conditions (Van Wagner 1977) such as ladder fuels provided by basal conifer branches (Hély et al. 2000). Accordingly, we found that landscapes with more spruce had higher odds of initiation, and those with more aspen had lower odds. Flannigan and Wotton (1991) suggest the duff layer sheltered under conifers is more flammable in comparison to duff in hardwood stands. Duff moisture content also varies between open and canopy-sheltered areas in conifer stands (M. Wotton, CFS, pers. comm.), this suggests that the relative importance of moisture/weather and fuel type is scale-dependent.

The decreased odds of initiation associated with recently burned forest may be explained by intense fires leaving lush, young trees or graminoid regeneration after fire. In mesic stands, early regeneration is typically dominated by aspen regardless of the subsequent developmental pathway. In xeric areas, burned jack pine stands typically regenerate to jack pine and aspen, the former as a result of seed release from serotinous cones (Greene et al. 2004). The inclusion of burned forest area in the models accounts for changes in the probability of

subsequent initiations in a landscape after a fire and demonstrates age-dependent initiation frequency. The years 1980-1982 (prior to the study period) and 1990 were extreme fire years in the region; roughly 13 660 km² of the study area (0.2%) was burned in 1990. The forest composition variable for burned area was not stratified by year-since-burn in our analysis. These stands would vary in age from one to 12 years old. Chen and Popadiouk (2002) suggest that lightning strikes themselves would be less frequent in shorter, young stands, but lightning is thought to propagate down randomly from a cloud without influence of objects on the ground until it is 50-100 m above ground when it chooses its target within a radius of about 100 m (M. Uman, U. of Florida, pers. comm.) suggesting that stand age as a function of height did not influence initiation patterns at the landscape scale of our study. The dampening of initiation with burned and deciduous area and increasing initiation with spruce area suggests the probability of initiation increases with stand age as a function of stand composition in the mixedwood boreal. Ignoring these age- (Van Wagner 1978; Boychuk et al. 1994) or species-initiation feedbacks in natural disturbance regimes could result in an incomplete or misleading synthesis of boreal dynamics and the potential influences of forest management.

We detected a quadratic relationship between initiation and black spruce composition suggesting that initiations were more likely in landscapes with intermediate amounts of black spruce. The relationship reached a maximum when 30 % compositional area of the landscape was black spruce-dominated. Since black spruce can occupy poorly drained sites this may reflect regional hydrology such that soil landscapes where black spruce is most abundant are also very moist and less conducive to initiation. It could be argued that soil moisture gradients play a strong role in determining stand composition so it would be more appropriate to compare the relative influence of two abiotic factors: soil landscapes and weather patterns, on fire initiation. Though this may be true at some spatial scales, the differences in forest composition effects estimated by our analyses were dominated by white spruce and aspen. Either of these species can each be expressed on the same mesic site through a range of development

pathways. Because of our coarse scale of analysis, detailed heterogeneity among and within stands or within cover types could not be addressed.

Initiation did not vary with the amount of jack pine in a landscape. This was surprising since *Pinus* is regarded as fire-adapted genus (Richardson 1998). Jack pine typically grows on rapidly draining soils with dry surface fuels readily flammable in match-ignited field trials (J. Beverly, CFS, pers. comm.). Perhaps ignition is successful in dry, fine fuels of a mature jack pine stand's understory but a dearth of coarse fuel for sustained combustion on the forest floor results in a proportionate initiation frequency. In Alberta, mature pine is considered a "go, no-go" fuel type (assuming ladder fuels such as a spruce understory are not present) where fire behaviour is considered low except under strong winds (C. Tymstra, ASRD, pers. comm.). This may be further enhanced through the mixing of pine and aspen in the mixedwood region. Initiation was not sensitive to the amount of open (e.g., muskeg) area in a landscape.

Weather, representing fuel moisture conditions explained substantial variation in initiation frequency demonstrated by a positive relationship with two JWL indices and SSR. The JWLs represented potential initiation conditions by combining appropriate fuel moisture levels and lightning activity. More variation was described by the JWLs than lightning and weather alone or together. The superior performance of transformed JWLs suggested the relationship with initiation was not additive but that initiation efficiency decreased at higher values of JWL. This non-additivity might be an artifact of the binary response data since we were simply modelling the probability of a single initiation per year per landscape. We expected that weighted JWLs would be better descriptors of initiation than JWLs since they represented a hypothesis that storms with high lightning density also produced more precipitation and would result in decreased initiation potential. Only one wJWL was selected in the model W_1 , and this term dropped out in WF_1 and WF_2 . The failure of the wJWLs might reflect a temporal disconnection between lightning and weather data. Weather data were collected at noon whereas lightning "days" were 24 hours in duration from midnight; errors in lightning detection are discussed in *Methods*. The positive relationship with SSR

indicated that the regional climate variability among years resulted in a change in initiation probability. The relationship with the JWLs and SSR were demonstrated at a voxel and regional scale, respectively, and together highlighted multi-scale influences of weather conditions on initiation.

Initiation probability increased with elevation despite a negative relationship between elevation and the JWL indices. In contrast, L_{COUNT} and L_{DAYS} increased with elevation though these lightning indices did not explain sufficient variation to be kept in the WF_1 or WF_2 model. The critical value for creating the JWLs was set at weather conditions where 85% of initiations were recorded so the range in conditions viable for initiation is not fully represented by the JWL indices. Initiation potential could increase with elevation through the integration of increased lightning activity and appropriate (yet sub-critical) fuel moisture codes. We had no evidence for a relationship between elevation and initiation through variation in fuel since elevation was not correlated with any forest types. Northing (latitude) was a potential proxy for growing season length or additional unmeasured gradients in climate since fewer initiations were recorded in more northerly landscapes even after accounting for variation in elevation, weather and forest composition.

Our best model, WF_2 , included fixed effects and a random effect for landscape. We examined the spatial distribution of random effects to suggest processes responsible for this stability of initiation within the landscape scale; nothing was apparent. We had detected a stable pattern in cumulative JWLs but these were structured at a much broader scale than a single landscape and were found to be a direct reflection of elevation. We did not include anthropogenic linear features such as roads and seismic lines, the distribution of mineral deposits nor metallic structures such as compressor stations or oil and gas wells which are abundant in the region (Schneider et al. 2003). It would be interesting to determine if these features could be responsible for local, stable variation of initiation or have an indirect influence on natural disturbance patterns in the boreal.

Our study is the first conclusive demonstration that variation in fuel type influences initiation patterns in the boreal forest. Though we only address the initiation phase of fire behaviour in our study, the ecological consequence implies that variable flammability of species regulates this natural disturbance endogenously. Recent studies have illustrated that the frequency and size of boreal fires are related to vegetation. Bergeron et al. (2004) demonstrated the influence of landscape composition on burn rate in the mixedwood and coniferous boreal forests of northwestern Quebec. The effectiveness of deciduous stands as fire breaks that limit fire size has been proposed in simulation models on Alberta landscapes (Hirsch et al. 2001), and preferential burning of coniferous forest stands has been shown in the compositional analysis of area burned in this same study area (Cumming 2001).

The relationship between fire, weather and forest composition may vary depending on heterogeneity, study scale, and the component of fire behaviour. The boreal forest is circumpolar and its structure varies. We would expect similar patterns to those we identified here in other mixedwood boreal regions such as Saskatchewan, Ontario and Quebec. Much of the more northerly and mountainous boreal regions are conifer-dominated and it is difficult to infer how sensitive initiation patterns would be to the decreased heterogeneity in landscape composition found in such regions. Where fuel type is less variable, initiation patterns will likely be more strongly associated with heterogeneity in fuel moisture/weather conditions for any ecosystem. Bessie and Johnson (1995) used data from sub-alpine conifer stands exhibiting relatively limited heterogeneity and demonstrate that fire behaviour was dominated by inter-annual weather variation rather than variation in fuel. Renkin and Despain (1992) demonstrated how the relationship with forest type varied between initiation and spread in the conifer-dominated region of Yellowstone National Park, and showed that the influence of forest cover type on fire activity decreased under drought conditions. However, we found that in more extreme fire weather years the detected effect of forest composition (amount of white spruce in a landscape) on fire initiation increased, as illustrated through a positive interaction between S_w and SSR. It is possible that

climate change may precipitate even more extreme fire weather that could overwhelm any endogenous, biotic regulation of initiation patterns in the mixedwood study region however we issue a caveat that changes in forest composition (and heterogeneity) through development, exploration, forestry, insect outbreaks could also trigger even more pronounced biotically-regulated patterns of initiation.

Many existing models of natural disturbance integrate empirically-derived stochastic initiation probabilities however estimates that allow spatio-temporal variation in environmental characteristics that influence fire initiation are critical to simulation models of ecosystem dynamics (He and Mladenoff 1999) since we (humans) insist on novel and dramatic alterations to our environment. We provide an example that answers a call to develop appropriate empirically based estimates from which to test ecological hypotheses (Yang et al. 2004). The statistical models used in this study provided parameter estimates relating initiation to biotic and abiotic sources; in this system these are endogenous and exogenous regulators to natural disturbance, respectively. Our study illustrates methods and results to enhance our understanding of weather-vegetation-disturbance interactions in boreal forest ecosystems and a framework to generate hypotheses about the integrated effects of ongoing changes in land cover, climate, and natural disturbance to ecosystem dynamics.

Table 2-1. Summary of variables used in logistic regression models.

Weather (fuel moisture)			
Index	Classification	Range (mean)	
FFMC _{DAYS}	number of days per voxel above threshold FFMC	0-44 (16)	
DMC _{DAYS}	number of days per voxel above threshold DMC	0-56 (14)	
SSR	regional annual seasonal severity rating of fire weather	0.8-1.4 (1.06)	
L _{DAYS}	number of days with lightning per voxel	0-29 (3)	
L _{COUNT}	sum of lightning strikes per voxel	0-332 (37)	
FFMC _{JWL}	joint FFMC – lightning index	0-12 (2)	
DMC _{JWL}	joint DMC – lightning index	0-13 (2)	
ISI _{JWL}	joint ISI – lightning index	0-15 (3)	
wFFMC _{JWL}	weighted FFMC _{JWL}	0-10.4 (1.4)	
wDMC _{JWL}	weighted DMC _{JWL}	0-12 (1.5)	
Geography			
ELEV	Elevation at the centroid of each landscape	227-1019 m	
NORTH	UTM at centroid of landscape	6 100 565 – 6 428 588	
Forest composition		Landscape	Regional
Stand cover type	Forest classification	Range (median) ^{&}	Proportion ^s (found) [*]
Deciduous (A _L)	deciduous canopy dominance and less than 25% canopy closure, up to 50% conifer by volume	0-1500 (27)	0.01 (71%)
Deciduous (A _H)	deciduous canopy dominance and greater than 25% canopy closure, up to 50% conifer by volume	0-8400 (2200)	0.27 (99%)
white spruce (S _w)	white spruce canopy dominance and up to 50% other species by volume	0-3600 (360)	0.06 (90%)

continued

Forest composition		Landscape	Regional
jack pine (P _J)	jack pine canopy dominance	0-6300 (460)	0.09
			(95%)
black spruce (S _B)	black spruce canopy dominance	0-6400 (540)	0.11
			(95%)
open (O)	low density scrub and all wetland types (not including water bodies)	100-9200 (3300)	0.38
			(100%)
burned (B)	area burned	0-9300	0.06
		(0)	(35%)
various [@]	burned or logged after inventory with unknown pre-disturbance cover type, cut at time of inventory, or unknown date of disturbance	0-7300 (0)	0.02

^sMean values from all 946 sample landscapes were used in regional proportions of forest composition since some values changed slightly over the study period as a result of disturbances.

* The percentage of landscapes wherein the forest type is found

& Area in ha per landscape

[@] Not included in analyses due to small sample sizes

Table 2-2. Parameter estimates (β) and standard errors (s.e.) from linear regression models used to validate interpolation of fire weather indices, FFMC, DMC and ISI.

Dependent variable	Independent variable	n	β	s.e.
1. interpolated FFMC	observed FFMC	20	0.96	0.063
residuals from 1.	elevation	20	-0.011	0.16
2. interpolated DMC	observed DMC	20	0.87	0.16
residuals from 2.	elevation	20	-0.010	0.18
3. interpolated ISI	observed ISI	20	0.84	0.071
residuals from 3.	elevation	20	0.0017	0.0016

Table 2-3. Selection and diagnostics for logistic regression models of initiation. WF_2 includes a random intercept for landscape. Table 2-4 summarizes parameter estimates of WF_2 . Appendix B provides descriptions of all variables.

Model	Model terms	ℓ	k	AIC	ROC	R^2
Null	intercept only (β_0)	3880	1	7762	NA	NA
W_0 global	$DMC_{DAYS} + FFM C_{DAYS} + FFM C_{JWL} + DMC_{JWL} + ISI_{JWL} + wFFM C_{JWL} + wDMC_{JWL} +$	3736	11	7495	0.64	0.052
weather	$SSR + L_{COUNT} + L_{DAYS}$					
W_1 best weather	$\ln(FFM C_{JWL}) + \ln(DMC_{JWL}) + wFFM C_{JWL} + SSR + L_{DAYS}$	3738	6	7489	0.64	0.051
F_0 global fuel	$B + A_L + A_H + S_W + S_B + S_B^2 + O + P_j$	3705	9	7429	0.65	0.063
F_1 best fuel	$B + A_L + A_H + S_W + S_B + S_B^2$	3706	7	7427	0.65	0.063
WF_0 global	$B + A_L + A_H + S_W + S_B + S_B^2 + wFFM C_{JWL} + \ln(FFM C_{JWL}) + \ln(DMC_{JWL}) + SSR + L_{DAYS} +$ $ELEV + NORTH$	3534	14	7097	0.71	0.12
WF_1 best	$B + A_H + S_W + S_B + S_B^2 + \ln(FFM C_{JWL}) + \ln(DMC_{JWL}) + SSR + ELEV + NORTH$	3537	11	7097	0.71	0.12
WF_2 best	$B + A_H + S_W + S_B + S_B^2 + \ln(FFM C_{JWL}) + \ln(DMC_{JWL}) + SSR + ELEV + NORTH$	3528	12	7081	NA	NA

ℓ is the negative of the log-likelihood from maximum likelihood estimation

k is the number of parameters estimated in the model

Table 2-4. Maximum likelihood estimates and their standard errors for each term in WF₂ (Table 2-3). ΔAIC is the increase of AIC when each term is removed from the mixed effects model.

Term	β	s.e.	ΔAIC
β_0	+7.03	2.74	
B	-2.85	0.41	58
A _H	-2.52	0.26	93
S _W	+4.40	0.57	55
S _B	+2.91	0.87	9
S _B ²	-4.74	2.10	3
ln(FFMC _{JWL})	+0.23	0.050	20
ln(DMC _{JWL})	+0.27	0.048	31
ELEV	+0.0012	0.00032	12
SSR	+1.26	0.21	35
NORTH	-0.0000018	0.00000042	16
s.d. of random effect: <i>landscape</i>		0.44	16

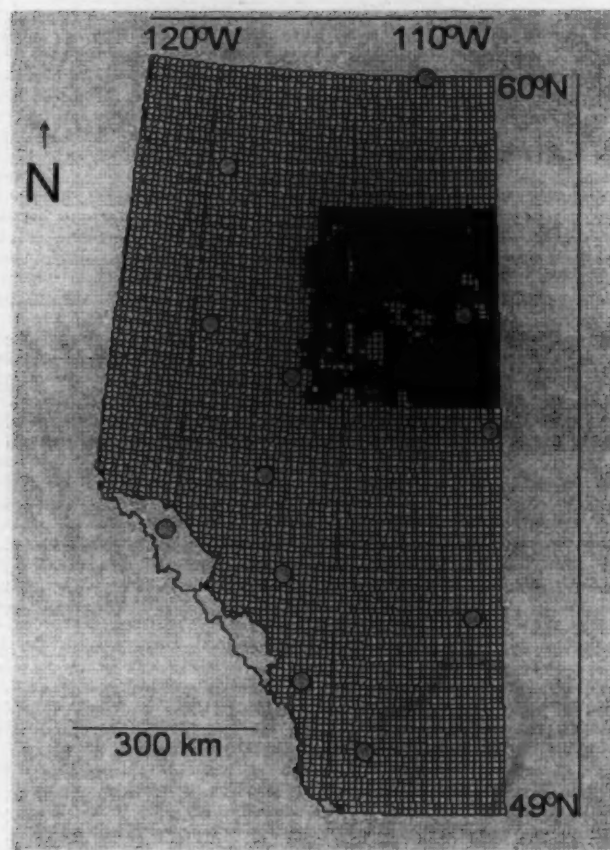


Figure 2-1. A map of the study region (black) within the Province of Alberta. Gridded lines show the spatial sample grain, 10 000 ha landscape units. Open circles show the location of lightning detection stations.

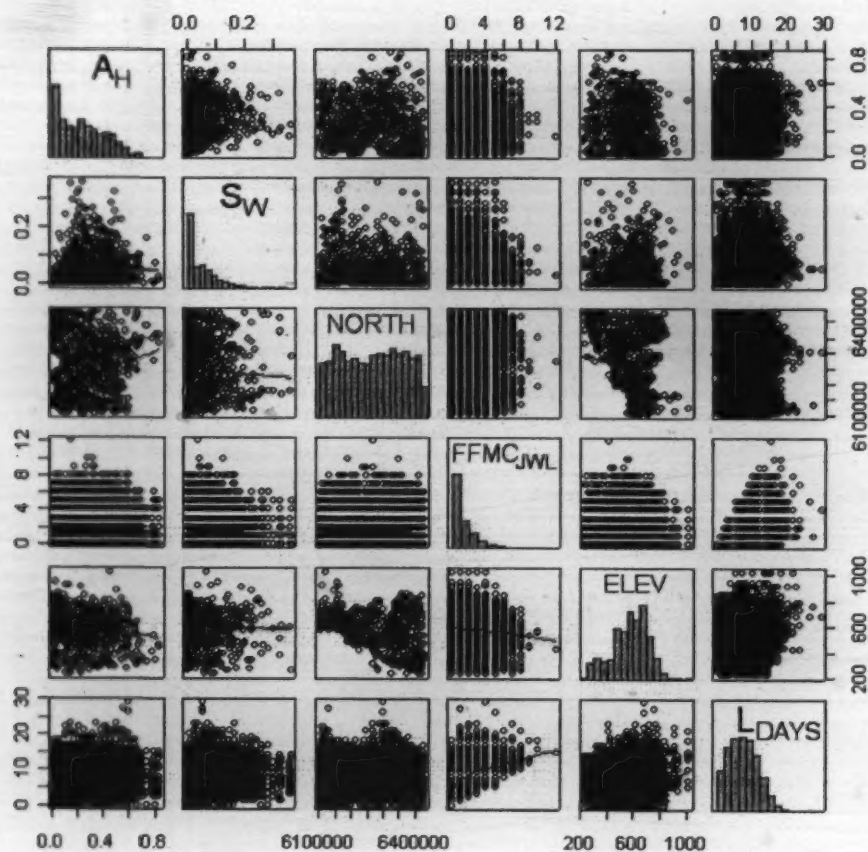


Figure 2-2. The inter-correlation and distribution of six study variables used in logistic regression analyses. From top left these include: that area of high density aspen dominated forest (A_H), white spruce dominated forest (S_W), spatial location based on UTM co-ordinates (NORTH), the Fine Fuel Moisture Code joint weather-lightning index ($FFM_{C_{JWL}}$), elevation and the number of days in the fire season when lightning activity was detected (L_{DAYS}).

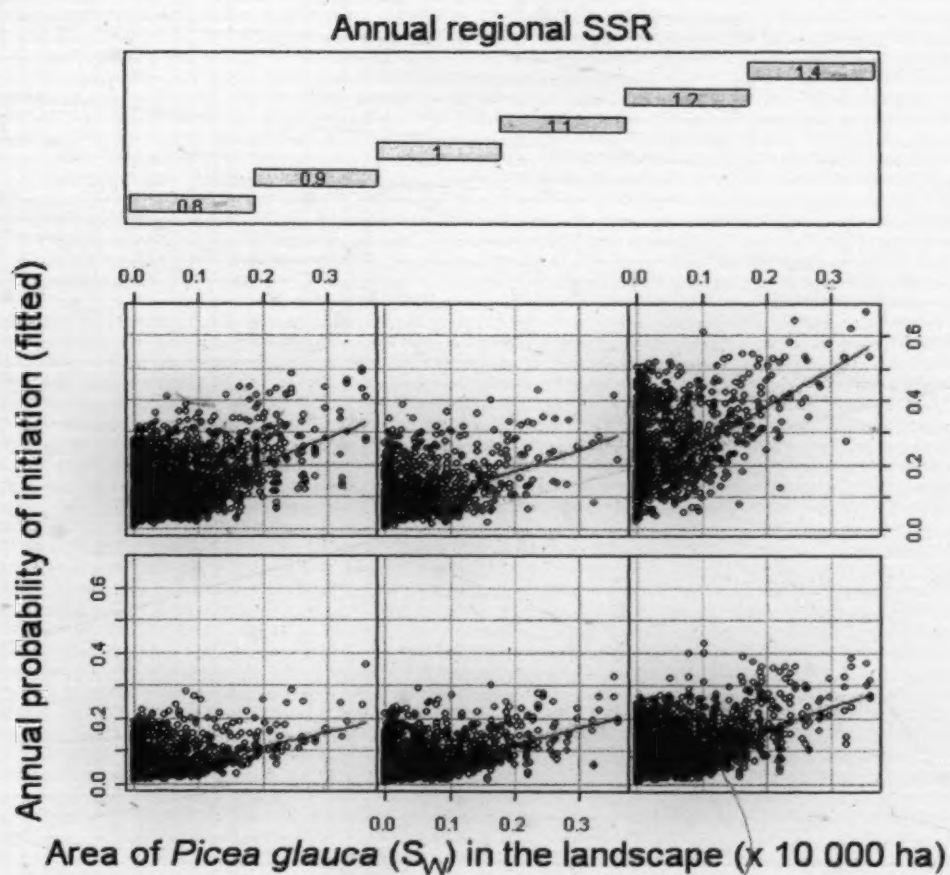


Figure 2-3. The increased effect of forest composition in years with more extreme fire weather illustrated with by the positive interaction between seasonal severity rating (SSR) and the area of white spruce (S_W) in the landscape.

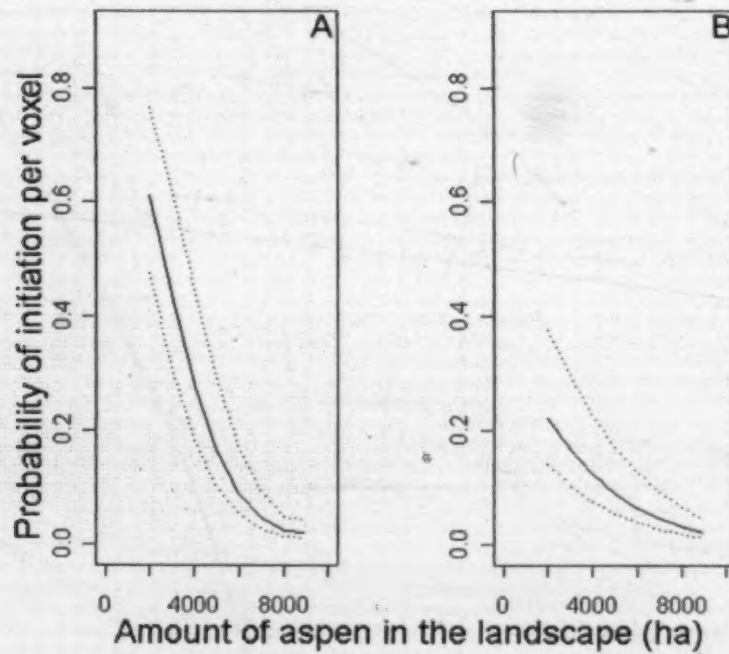


Figure 2-4. The probability of initiation per voxel varies with forest composition; spruce and aspen dominated landscapes promote opposing initiation frequencies. The solid black lines illustrate the expected probability of initiation as the amount of aspen increases and the amount of A) white spruce decreases or B) black spruce decreases. The dotted black lines show the interval of expected probabilities calculated from the central 95% of values observed for weather variables, $FFMC_{JWL}$ and DMC_{JWL} .

Literature Citations

- Albani, M., D.W. Andison, and J.P. Kimmins. 2005. Boreal mixedwood species composition in relationship to topography and white spruce seed dispersal constraint. *Forest Ecology and Management* **209**: 167-180.
- Anderson, K., and P. Englefield. 2001. Quantile characteristics of forest fires in Saskatchewan. Pages 9-16 in *Proceedings of the Fourth Symposium on Fire and Forest Meteorology*, 13-15 November, 2001, Reno Nevada. American Meteorological Society, Boston, Massachusetts.
- Anderson, K., D.L. Martell, M.D. Flannigan, and D. Wang. 2000. Modeling of fire occurrence in the boreal forest region of Canada. Pages 357-367 in E. Kasischke and B.J. Stocks, editors. *Fire, climate change and carbon cycling in the boreal forest*. Springer-Verlag, New York.
- Anonymous. 1985. Alberta Phase 3 Forest Inventory: Forest cover type specifications. Alberta Forestry, Lands and Wildlife, Forest Service ENR Report No. Dept. 58.
- Bergeron, Y., S. Gauthier, M.D. Flannigan, and V. Kafka. 2004. Fire regimes at the transition between mixedwood and coniferous boreal forest in northwestern Quebec. *Ecology* **85**:1916-1932.
- Bessie, W.C., and E.A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forest. *Ecology* **76**:747-762.
- Boyчук, D., A.H. Perera, M.T. Ter-Mikaelian, D.L. Martell, and L. Chao. 1997. Modelling the effect of spatial scale and correlated fire disturbances on forest age distribution. *Ecological Modelling* **95**:145-164.
- Broström, G. 2003. Generalized linear model with random intercepts. <http://www.stat.umu.se/forskning/glmmML.pdf>
- Burnham, K.P., and D.R. Anderson. 1998. *Model selection and inference: A practical information-theoretic approach*. Springer-Verlag, New York.
- Chen, H.Y.H., and R.V. Popadiouk. 2002. Dynamics of North American boreal mixedwoods. *Environmental Reviews* **10**:137-166.
- Cressie, N.A. 1991. *Statistics for spatial data*. John Wiley & Sons, New York.
- Cumming, S.G. 2001. Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn? *Ecological Applications* **11**:97-110.

- Cumming, S.G. 2005. Effective fire suppression in boreal forests. *Canadian Journal of Forest Research* 35:772-786.
- Cunningham, A.A., and D.L. Martell. 1976. The use of subjective probability assessments concerning forest fire occurrence. *Canadian Journal of Forest Research* 6:348-356.
- Diaz-Avalos, C., D.L. Peterson, E. Alvarado, S.A. Ferguson, and J.E. Besag. 2001. Space-time modelling of lightning-caused ignitions in the Blue Mountains, Oregon. *Canadian Journal of Forestry Research* 31:1579-1593.
- Didham, R.K., and C.H. Watts. 2005. Are systems with strong underlying abiotic regimes more likely to exhibit alternative stable states? *Oikos* 110:409-416.
- Fielding, A.H., and J.F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* 24:38-49.
- Flannigan, M.D., I. Campbell, B.M. Wotton, C. Carcaillet, P. Richard, and Y. Bergeron. 2001. Future fire in Canada's boreal forest: Paleoecology results and general circulation model - regional climate model simulations. *Canadian Journal of Forestry Research* 31:854-864.
- Flannigan, M.D., and B.M. Wotton. 1991. Lightning-ignited forest fires in northwestern Ontario. *Canadian Journal of Forestry Research* 21:277-287.
- Flannigan, M.D., and B.M. Wotton. 1989. A study of interpolation methods for forest fire danger rating in Canada. *Canadian Journal of Forestry Research* 21:66-72.
- Fowler, P.M., and D.O. Asleson. 1984. The location of lightning-caused wildland fires, northern Idaho. *Physical Geography* 5:240-252.
- Fuquay, D.M., R.G. Baughman, and D.J. Latham. 1979. A Model for Predicting Lightning Fire Ignitions in Wildlands Fuels. Research Paper INT-217. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah.
- Greene, D.F., J. Noël, Y. Bergeron, M. Rousseau, and S. Gauthier. 2004. Recruitment of *Picea mariana*, *Pinus banksiana*, and *Populus tremuloides* across a burn severity gradient following wildfire in the southern boreal forest of Quebec. *Canadian Journal of Forestry Research* 34:1845-1857.
- Gunderson, L.H., and C.S. Holling. 2002. *Panarchy: Understanding transformations in human and natural systems*. Island Press, Washington, D.C.

- Hartford, R.A. 1989. Smoldering combustion limits in peat as influenced by moisture, mineral content, and organic bulk density. Pages 282-286 in D.C. MacIver, H. Auld, and R. Whitewood, editors. *Proceedings of the 10th Conference on Fire and Forest Meteorology*, Ottawa, Ontario. AES, Downsview, Ontario, Canada.
- He, H.S., and D.J. Mladenoff. 1999. Spatially explicit and stochastic simulation of forest-landscape fire disturbance and succession. *Ecology* **80**: 81-99.
- Hély, C., M.D. Flannigan, Y. Bergeron, and D. McRae. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. *Canadian Journal of Forestry Research* **31**:430-441.
- Hély, C., Y. Bergeron, and M.D. Flannigan. 2000. Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. *Journal of Vegetation Science* **11**:813-824.
- Hirsch, K., V. Kafka, and B. Todd. 2001. Using forest management techniques to alter forest fuels and reduce wildfire size: An exploratory analysis. in R.T. Engstrom, and W.J. de Groot, editors. *Proceedings of the 22nd Tall Timbers Fire Ecology Conference: Fire in Temperate, Boreal, and Montane Ecosystems*. Tall Timbers Research Station, Tallahassee, FL.
- Hosmer, D.W., T. Hosmer, S. Lemeshow, S. le Cessie, S. Lemeshow. 1997. A comparison of goodness-of-fit tests for the logistic regression model. *Statistics in Medicine* **16**:965-980.
- Kourtz, P.H., and J.B. Todd. 1991. Predicting the daily occurrence of lightning-caused forest fires. *Forestry Canada Information Report PI-X-112*. Petawawa National Forestry Institute.
- Latham, D.J., and J.A. Schlieter JA. 1989. Ignition Probabilities of Wildland Fuels Based on Simulated Lightning Discharges. Research Paper INT-411. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.
- Legendre, P., and L. Legendre. 1998. *Numerical ecology*. 2nd English edition. Elsevier, Amsterdam.
- Legendre, P., M.R.T. Dale, M-J. Fortin, P. Casgrain, and J. Gurevitch. 2004. Effects of spatial structures on the results of field experiments. *Ecology* **85**:3202-3214.
- Mantel, N. 1967. The detection of disease clustering and a generalized regression approach. *Cancer Research* **27**:209-220.

- McCullagh, P., and J.A. Nelder. 1989. Generalized Linear Models. 2nd edition. Chapman and Hall, London.
- Nash, C.H., and E.A. Johnson. 1996. Synoptic climatology of lightning-caused forest fires in subalpine and boreal forests. *Canadian Journal of Forestry Research* 26:1859-1874.
- Nimchuk, N. 1989. Ground truthing of LLP lightning data in Alberta. Pages 33-40 in D.C. MacIver, H. Auld, and R. Whitwood, editors. Proceedings of the 10th Conference on Fire and Forest Meteorology, Ottawa, Ontario. AES, Downsview, Ontario, Canada
- Paine, R.T., and S.A. Levin. 1981. Intertidal landscapes: Disturbance and the dynamics of pattern. *Ecological Monographs* 51:145-178.
- Podur, J., D.L. Martell, and F. Csillag. 2003. Spatial patterns of lightning-caused forest fires in Ontario 1976-1998. *Ecological Modelling* 164:1-20.
- R Development Core Team. 2005. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>
- Rakov, V.A., and G.R. Huffines. 2003. Return-stroke multiplicity of negative cloud-to-ground lightning flashes. *Journal of Applied Meteorology* 42:1455-1462.
- Renkin, R.A., and D.G. Despain. 1992. Fuel moisture, forest type, and lightning-caused fire in Yellowstone National Park. *Canadian Journal of Forestry Research* 22:37-45.
- Richardson, D.M. 1998. Ecology and Biogeography of *Pinus*. Cambridge University Press.
- Rothermel, R.C. 1972. A mathematical model for predicting fire spread in wild land fuels. USDA Forest Service, Research Paper INT-115.
- Rowe, J.S. 1972. Forest regions of Canada. Publication No. 1300, Canadian Forest Service, Ottawa, Ontario.
- Schneider, R.R., J.B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the Western Canadian Sedimentary Basin: A modelling approach. *Ecology and Society* 7(1):8.[online].
[URL:http://www.ecologyandsociety.org/vol7/iss1/art8](http://www.ecologyandsociety.org/vol7/iss1/art8)

- Schoennagel, T., T.T. Veblen, and W.H. Romme. 2004. The interaction of fire, fuels and climate across Rocky Mountain forests. *Bioscience* **54**:661-676.
- Stocks, B.J., J.A. Mason, J.B. Todd, E.M. Bosch, B.M. Wotton, B.D. Amiro, M.D. Flannigan, K.G. Hirsch, K.A. Logan, D.L. Martell, and W.R. Skinner. 2003. Large forest fires in Canada, 1959-1997. *Journal of Geophysical Research*. **108**(D1).
- Strong, W.L., and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands, and Wildlife, Land Information Division, Resource Information Branch, Edmonton, AB.
- Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. *Canadian Journal of Forestry Research* **7**:23-34.
- Van Wagner, C.E. 1978. Age-class distribution and the forest fire cycle. *Canadian Journal of Forestry Research* **8**:220-227.
- Van Wagner, C.E. 1987. Development and structure of the Canadian Forest Fire Weather Index System. *Can. For. Serv., Ottawa, Ont. For. Tech. Rep.* 35.
- Wierzchowski, J., M. Heathcott, and M.D. Flannigan. 2002. Lightning and lightning fire, central cordillera, Canada. *International Journal of Wildland Fire* **11**:41-51.
- Yang, J., H.S. He, and E.J. Gustafson. 2004. A hierarchical fire frequency model to simulate temporal patterns of fire regimes in LANDIS. *Ecological Modelling* **180**:119-133.

Chapter 3. Evidence for accelerated, compound disturbance in the mixedwood boreal forest via increasing lightning fire initiation in harvested landscapes*

Introduction

The emulation of natural disturbance regimes as a coarse-scaled strategy for sustainable ecosystem management (Hunter 1993) is a prominent forest practice and policy in Canada's boreal region (Paine et al. 1998; McRae et al. 2001; Song 2002; Armstrong et al. 2003). Where fire is the dominant natural disturbance, historical wildfire patterns and forest age structure are used to estimate parameters of the fire regime, such as the frequency (Reed 2001), or size (Cumming 2001) of fires. Harvest prescriptions are then guided by derived indices such as burn rates, patterns of forest structure (patch size, stand composition and configuration) produced by fires, and the residual material left post-fire that reflect the natural range to which the ecosystem is adapted (Hunter 1993; McRae et al. 2001; Gillson and Willis 2004). The success of forest harvest in emulating natural disturbance has been rigorously studied in Alberta's mixedwood boreal region (Song 2002) where large, stand-replacing fires initiated by lightning strike are the dominant natural disturbance (Alberta Sustainable Resource Development [ASRD] 2004). Short-term comparisons between fire and harvest treatments for ecological indicators such as soil properties, forest regeneration, bird communities and landscape structure show that current harvest practices have a limited ability to emulate fire at some spatial and temporal scales (Song 2002). There has been little information collected or synthesized to provide evidence for how well harvesting emulates fire *for* fire – i.e., the influence of harvesting on future disturbance events. Disturbance events are an important agent in many systems and resilience theory suggests that changes to key ecosystem properties can increase the likelihood of a subsequent disturbance event causing a sudden shift to a new system state (Scheffer et al. 2001;

* A version of this chapter has been submitted for publication. Krawchuk and Cumming, *submitted*. Ecological Applications.

Gunderson and Holling 2002). It is imperative to determine how harvesting interacts with subsequent fire events to understand long term effects of forest management on boreal systems.

The relative influence of fuel type and weather /fuel moisture conditions on fire behaviour is controversial; both factors contribute to fire behaviour and regime at different scales in different systems (Renkin and Despain 1992; Bessie and Johnson 1995; Hély et al. 2001; Schoennagel et al. 2004; Bigler et al. 2005; Krawchuk et al. 2006). In mixedwood boreal forests, studies have demonstrated that forest type influences burn patterns (Cumming 2001a), area burned (Hirsch et al. 2001), fire initiation (Krawchuk et al. 2006), fire behaviour (Quintilio et al. 1991) and fire management efficacy (Arienti et al. 2006). These effects can be explained in part by variation in the properties of surface fuel available in coniferous, mixed and deciduous stands of the mixedwood region (Hély et al. 2000; Krawchuk et al. 2006) that varies as a function of regeneration time and disturbance frequency (McIntire et al. 2005). In managed landscapes, fuel heterogeneity is generated by harvest as well as burn activity. Alexander (1982) and Baxter (2004) suggest that forest harvesting influences fire hazard in the short-term through the piling of logging debris (tops, limbs, un-merchantable stems and logs) and exposing debris to atmospheric dessication (Kiil 1968). In contrast, recently burned areas have depressed re-burn potential that has been shown to persist for up to three decades, depending on fire intensity (Schiminel and Granstrom 1997; Lavoie 2004). In the mixedwood boreal forests of Alberta, Canada, fire operations personnel rate harvested stands as a slash fuel type (Forestry Canada Fire Danger Group 1992) for the first few years after logging, then as a grassy fuel type (Forestry Canada Fire Danger Group 1992) with high fuel load for up to 20 years (Kurt Frederick, ASRD, pers. communication). Simulation experiments in western Alberta suggested harvesting might reduce the hazard of crown fire in the long-term through increased area of deciduous dominated stands, but did not account for short-term increases in post harvest fire hazard from increased fine fuel loads in recently harvested stands (Hirsch et al. 2001). Otherwise, little has been done to quantify how harvesting influences

subsequent fire events in the boreal in the context of emulating natural disturbance. We are not aware of any studies that examine how harvesting affects the spatiotemporal pattern of lightning fire initiation, nor the long-term or regional implications this may have to disturbance and forest dynamics. Systematic changes in the location of fire events attributable to forest management could have substantial effects on the fire regime in the mixedwood boreal forest.

The objective of our study was to assess the relationship between lightning fire initiation and landscape composition under a mixed disturbance regime in the mixedwood boreal forest region of Alberta, Canada. We were specifically interested in comparing the probability of initiation in areas that had been recently burned relative to areas recently harvested. We quantified the annual pattern of initiation within $\approx 10\,000$ ha landscapes over eight years (1994 to 2001) in a $58\,000\text{ km}^2$ study region with respect to spatiotemporal variation in forest composition, lightning and fire weather conditions. We then used the stochastic simulation model, Tardis (Cumming and Armstrong 2004) to characterize potential long-term (100 years) effects of the mixed disturbance regime (wildfire and forest harvest) and used indicators of return interval and area affected to quantify the altered disturbance frequency of the region.

Methods

Study region

The study region was a sample of mixedwood boreal forest (Rowe 1972) in central-eastern Alberta, Canada. The regional climate is continental with mean winter and summer temperatures of -13.2°C and $+13.5^\circ\text{C}$, respectively and total precipitation in summer and winter in the range of 240 mm and 64 mm, respectively (Strong and Leggat 1992). There is limited relief with elevation varying from 200 to 1000 m. Glaciation resulted in a surficial geology of rolling moraine deposits on the uplands and flat glacio-lacustrine deposits on the lowlands (Rowe 1972). The mixedwood is the most productive and diverse of boreal ecosystems in North America (Chen and Popadiouk 2002). Forest tree communities are stratified by soil moisture regime (Strong and Leggat 1992).

Upland mesic sites are dominated by mixtures of trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.) or white spruce (*Picea glauca* (Moench) Voss). Post-fire community development at these sites depends on variation in disturbance history, composition, and spatial context (e.g., stochastic availability of propagules from neighbouring stands; (Albani et al. 2005)). Paper birch (*Betula papyrifera* Marsh) is present but pure stands are rare. White spruce and balsam fir (*Abies balsamea* (L.) Mill) dominate older stands but pure stands are not currently abundant. Xeric sites are vegetated by jack pine (*Pinus banksiana* Lamb), sometimes in association with aspen. Moist, poorly drained sites are vegetated by black spruce (*Picea mariana* (Mill.) B.S.P.). There are extensive wetlands including open areas of bog, fens, swamps and marshes. Tamarack (*Larix laricina* (Du Roi) Koch) is the dominant tree species in these wetlands.

The region has a mixed disturbance regime that includes fire and forest harvest. The area burned by lightning- and human-caused fires has contributed to the heterogeneous species and age structure of the region. Lightning fires are responsible for the majority of the area burned but greater access is increasing the frequency of human-caused fires. Annual area burned by lightning fires in the study area between 1961 and 1995 was described as a log-normal distribution by Armstrong (1999), ranging between 4 and 708 172 ha. Extensive commercial harvest of aspen in the region began in 1993. Small-scale conifer harvest has been ongoing since the 1950s. The subsequent details of forest operations were summarized from the forest management plan for the region (Alberta-Pacific Forest Industries, Inc. 2004) and Forest Stewardship Council certification documents (SmartWood 2005). The harvest prescription for the deciduous and coniferous forest types is clear-cut harvesting with minimum retention of merchantable in-block trees. The dominant harvest systems are feller-bunchers and grapple skidders. Current practices utilize road-side processing leaving the majority of slash along road-sides (Baxter 2004). Slash burning is not practiced (Baxter 2004). Renewal of deciduous sites relies on natural regeneration (suckering). Coniferous sites are typically planted to white spruce or pine. Mesic,

mixedwood sites are either planted or left for natural regeneration, the decision based on a multi-level forest regeneration policy (Cumming and Armstrong 2001). The deciduous harvest is composed of 99% poplar (aspen and balsam), the coniferous harvest is 62% white spruce, 30% jack pine and 8% black spruce, with a cumulative annual area harvested at approximately 14 000 ha. There is also substantial oil and gas development in the region resulting in overstory and understory removal from lines cut during seismic exploration (Schneider et al. 2003). Though the cumulative area cut for seismic is quite high, each linear feature is two to seven metres wide. While the influence of these linear features on fire behaviour is still debated, Arienti et al. (2006) show that the initial growth and suppression of fires is not influenced systematically by linear features. The resulting study region is composed of vegetation communities at various ages that were initiated through natural (lightning caused fire) and anthropogenic (harvest or human caused fire) disturbance. Table 3-1 reports the area burned and harvested in the study region between 1994 and 2001.

The Province of Alberta is partitioned using an administrative grid system of townships each measuring approximately 9 600 ha but varying slightly due to geographic corrections. Each township in the study region was considered as a spatial sampling unit, hereafter referred to as a landscape. The study region was divided into three-dimensional space-time samples (voxels) with the units landscape and year (Krawchuk et al. 2006). Data from 1994 and 2001 were collected from the fire season (May 1 to Sept 30) for 588 landscapes and summarized for each year providing 4704 voxels. All spatial data were analyzed in ArcGIS 9.1 (ESRI Inc. 2005) using the NAD27 UTM zone 12 projection. Statistical analyses were performed in R (version 2.2.0, R Development Core Team 2005).

Empirical data

1. Lightning fire initiation. We defined a lightning fire initiation as an ignition that becomes large enough to be detected. Fire initiation data were provided by Alberta Sustainable Resource Development (ASRD 2006). Un-

detected or un-recorded fires may have occurred in the region but fire detection efficiency appears to have been constant over the study interval (Cumming 2005). We collated fire data to describe the number of days per voxel wherein at least one lightning fire was detected (Table 3-2). Multiple fires within the same landscape on the same day were a result of highly correlated biophysical conditions and were counted as a single event. If one of these fires spread to a significant size, they would likely coalesce so the ecological implication of this aggregation is minimal. We refer to these counts as 'initiations'. Table 3-2 provides an overview of the initiation data.

We tested for spatial autocorrelation in the initiation data using Moran's I statistic (Legendre and Legendre 1998) to assess *a priori* if the assumptions of independent distribution in errors might be valid for count regression analysis of these data. Annual test statistics were calculated separately for the distances 0 to 10 km, 10 to 20 km, etc..., up to 60 km. We applied a Bonferroni, and progressive Bonferroni correction to the tests of significance of the Moran's I statistic (Legendre and Legendre 1998) with initial critical $\alpha = 0.05$. Autocorrelation was detected up to 20 km (2 landscapes) using progressive Bonferroni tests (11 of 16 test statistics [8 years x 2 distance classes = 16]), but Bonferroni tests suggested no autocorrelation was present (one of 16 test statistics).

2. Joint weather-lightning indices (JWLs). Lightning fire initiation requires the co-occurrence of a lightning strike and low fuel moisture content. An inverse relationship between lightning efficiency (number of fires/number of strikes) and fuel moisture (Nash and Johnson 1996) supports the notion that a co-occurrence of 'appropriate' daily fire weather conditions and daily lightning should better explain the distribution of initiations than the independent distributions of each. In previous work we found that JWLs outperformed independent measures of lightning or fire weather indices in predicting lightning fire occurrence (Krawchuk et al. 2006). We used two JWLs to describe the sum number of days when fuel moisture conditions were receptive to ignition and lightning was detected in each voxel.

The JWLs used spatially referenced daily lightning and fire weather data (Krawchuk et al. 2006). Lightning data were obtained from ASRD detection stations across the province and assessed using methods in Krawchuk et al. (2006). We summarized conditions relevant to fire ignition provided by The Canadian Forest Fire Weather Index System (FWI; Van Wagner 1987) to produce two fire weather indices for each voxel: the Fine Fuel Moisture Code (FFMC) and the Duff Moisture Code (DMC). The FFMC and DMC represent moisture content of litter or fast-drying fine fuels and loosely compacted organic layers of moderate depth, respectively. These codes were calculated from daily measurements of temperature, relative humidity, and precipitation taken from an array of 65 provincial weather stations operated and calibrated by ASRD in and around the region. The two daily codes were interpolated at a resolution of 1 km on each day of the fire season over 8 years by inverse-distance weighting (Flannigan and Wotton 1989) using the six nearest weather stations as informative neighbours (see Krawchuk et al. (2006) for validation).

We estimated a critical value where fuel moisture was 'receptive' to fire propagation by assessing the cumulative distribution of the recorded daily FFMC and DMC values for each initiation in a data set covering the same region over the time period 1983 to 1993 (Krawchuk et al. 2006). The critical value was that above which 85 % of initiations were detected. Threshold values of receptive fire weather calculated for FFMC and DMC were 87 and 34, respectively. These values agreed closely with analyses from a similar forest region in Saskatchewan, Canada (Anderson and Englefield 2001). We tallied the sum number of days per voxel when a given code exceeded the critical value and lightning was detected to create two joint lightning weather indices: $FFMC_{JWL}$, DMC_{JWL} (Table 3-3). Nimchuk (1989) estimated spatial error of Alberta's lightning detection system at 4 – 14 km. This error is larger than the spatial resolution of each landscape. In Krawchuk et al. (2006) we illustrated that daily lightning activity was spatially structured within years across the study region such that the influence of positional inaccuracy for strikes would be buffered by the spatial extent of storm activity with respect to the resolution of our study voxels.

3. Climate. We used the seasonal severity rating (SSR) from the FWI system (Van Wagner 1987) to measure annual variation within the climate of the region. The SSR was calculated as the mean annual Daily Severity Rating of all weather stations in the region over the fire season (Table 3-3). Severity ratings measure the control difficulty of a fire, based on wind speed, rainfall, humidity, and temperature (Van Wagner 1987); the SSR is used to describe fire characteristics of the year. There is a strong relationship between annual SSR (Table 3-3) and annual area burned (Table 3-1).

4. Forest fuel composition. We used the Alberta Vegetation Inventory (AVI) to measure fuel type composition in each voxel. The AVI database maps vegetation structure (forest canopy species composition, density), age, and disturbance history of forested and non-forested stands/areas at a 2 ha resolution interpreted from 1:15 000 air photo interpretation and field checking. The photography for our study was collected between 1992 and 1999. The AVI records the type and year of past disturbances – these attributes are considered accurate over the study interval. Thus, the forest composition of each landscape and each year can be estimated (Tables 3-1 and 3-4). Our data were updated with fire and harvest disturbances to 2001.

We adapted an existing ecological classification of the AVI (Cumming and Vernier 2002) to describe the multivariate fuel structure of each AVI polygon with 23 cover types (Table 3-4) for each year. The algorithm used the following information to classify each AVI polygon: the first to fifth dominant species and the associated percentage within each polygon according to crown closure; estimated year of stand origin; natural but non-forested cover types; anthropogenic cover types; disturbance type, year and extent. For each voxel, we tracked the area classified as harvested (H) and burned (B) in decadal classes (e.g., Table 4: B₁₀, B₂₀, B₃₀) for up to 30 years post disturbance. Polygons disturbed greater than 30 years prior were classified by stand characteristics. Our classification system for burned and harvested areas was consistent with the Fire Behaviour Prediction classifications estimated by fire operations personnel in Alberta (Kurt Frederick, ASRD, personal communication) and literature

comparing the structure of stands initiated by fire and harvest (Song (2002), and see *Introduction*) suggesting that burned and harvested stands convergence in structure over time. Harvested polygons were classified as harvested conifer (HC) or harvested deciduous (HD) based on AVI indication of post harvest silviculture treatment. We separated the harvest types into the two categories to capture potential variation in fuel type that might be caused by differences in harvest slash resulting from needle/leaf, branch and bark flammability, and in post-harvest regeneration. Harvesting occurs mostly in mid-winter or summer when the ground is frozen or reasonably dry, depending on the site. Fire occurs in the spring, summer and fall. It was not feasible to determine if fire initiations occurred before or after these disturbance treatments in a particular voxel. For example, a 1992 harvest could have occurred pre- or post-fire season. To avoid misclassification, the area of a landscape modified in a specific year was classified as harvested in (H_0) or burned in (B_0) (Table 3-4). Disturbance history attributes were not reliably recorded pre-1970. Our study frame begins in 1994 resulting in the "tail" years of burns and harvest (1965-1969) being unidentifiable for the oldest disturbance categories, B_{30} or H_{30} (e.g., for the study years 1994 to 1998). The information lost from these omissions was considered minimal.

The AVI overwrites original stand attributes when disturbance occurs. For example, a 1998 burn would be classified as B_0 for 1998, and B_{10} for the years 1999 to 2001. However the class for previous years (1994 to 1997) is unknown. Thus, in 1994, three percent of the study area had unknown composition, with the amount decreasing through time to near zero in 2001. Though the area affected was relatively small, this non-stationarity might influence results since some cover types covered small proportions of the region (Table 3-4). To account for this unknown component of the region, we developed post-processing algorithms to estimate the previous cover type of these polygons, one for harvested areas and one for burned areas. We used a coarse- and a fine-scaled backdating algorithm, and validated them with known data. The coarse-scaled method was aspatial and used the known composition of affected landscapes available from a previous version of forest inventory produced in 1983 to classify affected areas. The fine-

scaled method was spatially explicit and used two editions of the AVI to classify affected areas. The AVI was initially released with up-to-date disturbance information to 2000 – we refer to this as the OLDAVI. An updated edition was released with disturbance information up to 2001 - we refer to this as the AVI. The fine-scaled method used H-Tools (Beyer 2004) to classify affected cells in a rasterized image of each landscape by assigning them the cover type of their nearest spatial neighbour. Further details are available upon request. The validation suggested that both coarse- and fine-scaled methods would be appropriate to backdate landscape composition. We selected the coarse-scaled method since it was more efficient, equally suitable and the configuration of cover types provided by the fine-scaled method was not required for this study.

4. Geography. The elevation of each voxel was measured by its centroid, calculated from a mosaic of digital elevation maps (GEOBASE (www.geobase.ca)). The source digital data was extracted from the 1:250 000 scale National Topographic Data Base (Government of Canada, Natural Resources Canada, Earth Sciences Sector, Geomatics Canada, Centre for Topographic Information).

Exploratory data analyses

The variance/mean ratio of the initiation counts per voxel was 1.33 (Table 3-2) suggesting overdispersion relative to a Poisson distribution. Both the Akaike's Information Criterion (AIC), and the likelihood ratio test ($\alpha = 0.05$) between a negative binomial (NB) and Poisson (P) distribution suggested a NB model was most appropriate. The initiation data contained a large proportion (85%) of zero values which could explain the overdispersion so we also evaluated a zero-inflated Poisson (ZIP) and zero-inflated negative binomial model (ZINB) model with a logit link function on the binomial component of the model. We selected the zero-inflated mixture distribution over the zero-inflated conditional hurdle distribution based on our *a priori* understanding of the processes responsible for fire initiation (Martin et al. 2005). The mixture distribution consists of two parts, the binomial and negative binomial (or alternatively,

Poisson). Since a lightning fire event requires (by definition) appropriate weather events to occur, the binomial contribution to the overall $\Pr(Y=0)$ that is responsible for the zero-inflation might appropriately be described by weather/moisture variables. The negative binomial contribution to the overall count distribution would be described by weather/moisture, cover type, and geography. The AIC and Vuong statistic (based on the comparative predictive probabilities of non-nested models) suggested that a ZINB model of the raw count data was equally suitable to the NB, and these were superior to the Poisson and ZIP.

We assessed the bivariate relationship between initiation counts and all candidate covariates (JWLs, forest composition, geography) with generalized additive models using a Poisson family to assess the adequacy of a log-linear functional response. The composition variable BS (Table 3-4) was non-linear and the addition of a quadratic term BS^2 improved the fit substantially. We then fit generalized linear models with negative binomial error to estimate the polarity and magnitude of each relationship independently. The most striking trend was an opposite polarity between harvested and burned categories of all ages.

Highly correlated pairs of variables were screened using Pearson correlation coefficients. As expected, some burn (B) and harvest (H) categories presented in Table 3-4 (See * and @) resulted in high correlation (up to 0.8). To reduce the number of correlated variables we determined the most parsimonious partitioning for the H and B covariates separately by AIC with a negative binomial generalized linear model. We used an all-subsets selection method from a global model including all covariates for each group (H or B). The binning of six harvest ($HD_{(10,20,30)}$ and $HC_{(10,20,30)}$) variables to H_{ALL} , and of three burn ($B_{(10,20,30)}$) variables to B_{ALL} (Table 3-4) resulted in the most parsimonious description of fire initiation. The collapse of nine covariates to two resulted in a loss of information describing the time since disturbance however all the covariate patterns contributing to H_{ALL} were positive in bivariate analyses and all those contributing to B_{ALL} were negative suggesting that the procedure was not obscuring any age-related differences in polarity within each group. H_{ALL} and

B_{ALL} were uncorrelated with a Pearson correlation of $r = -0.054$. The variable HD_{10} explained the most variability within H and was also the variable with the greatest area (Table 3-4); the variable HC_{30} ranked second. There was also a high correlation (0.6) between $FFMC_{JWL}$ and DMC_{JWL} but we had no reason to maintain one over the other so we included both variables in the global regression model.

Regression analysis

We fit global regression models of fire initiation with 12 covariates. The variables included in the binomial component of the mixture model included: $FFMC_{JWL}$, DMC_{JWL} , SSR (Table 3-4). The variables included in the NB/P component of the model included: $FFMC_{JWL}$, DMC_{JWL} , SSR , H_{ALL} , B_{ALL} , MC , MD_{90+} , MD , JP , BS , BS^2 and $ELEVATION$ (Table 3-4). The NB model with all covariates remained superior to the P model using a likelihood ratio test ($\alpha = 0.05$), and the Vuong statistic suggested the ZINB was equally suitable to the NB. We adopted the ZINB as the final model because it allowed partitioning of covariates between those influencing the occurrence of any event and those influencing the counts of events.

We used a backward stepwise selection procedure from a global ZINB model to the final, best model using AIC. In the final model we tested the statistical significance of parameter estimates using the Wald statistic ($\alpha = 0.05$) in the spirit of pluralism between frequentist and information theoretic methods. After selecting the best model, we then assessed whether UTM measures of latitude and longitude explained any remaining variation in the data. We assessed whether the best model was still specified appropriately by the ZINB in comparison to the ZIP, NB and/or P error distributions using the Vuong test and likelihood ratio tests. We tested for a significant interaction between year, H_{ALL} and B_{ALL} to determine if these relationships were consistent across the relatively short study interval.

Pearson residuals were used to assess the best model using a-spatial plots, spatial and temporal variograms. We assessed the fit of the final model by the

difference between observed and predicted probabilities of counts using methods described by Lambert (1992). Model specification and goodness of fit were assessed using 1000 iterations of a parametric bootstrap (Efron and Tibshirani 1998) of the sum of squared errors of ZINB random deviates generated from the best model.

Simulation

We performed a simulation experiment to examine the long-term (100 years) sensitivity of forest system dynamics to the effect of harvested area on initiation. We used the simulation model, Tardis (Cumming and Armstrong 2001, 2004), an existing spatially explicit model of mixedwood boreal forest dynamics and forest planning developed and calibrated for this region. Tardis is a landscape resolution ($\approx 10\,000$ ha) simulator that includes forest harvest, table-driven stand dynamics, and a stochastic process of burning (Figure 3-2). Tardis operates at a time step of five years. Landscapes are described by an aspatial stand lists abstracted from AVI data. Burning of stands is simulated stochastically using a four-stage process of initiation, escape, final size, and effect (Cumming and Armstrong 2004). Harvest of forest stands is simulated by greedy optimization within each time step (Cumming and Armstrong 2001) and is deterministic conditional on the outcome of the fire simulation; Tardis does not provide intertemporal optimization. Cost equations and scheduling rules are based on the mixed-tenure (softwood and hardwood) planning environment that exists in the region. The growth, yield, succession and regeneration of forested stands were based on management models (Cumming and Armstrong 2004). A full description of Tardis can be found in Cumming and Armstrong (2004).

We implemented Tardis using model software and parameters as described in Cumming and Armstrong (2004), except we replaced the fire initiation module with our best model (Table 3-5). Our model differs from that used by Cumming and Armstrong (2004) in three ways. We include an explicit relationship between burned and harvested landscape composition and initiation, incorporate weather-related variables (SSR and $FFMC_{JWL}$), and use an annual

temporal resolution. Annual spatio-temporal variation of weather conditions was provided by the annual weather data from 1994 to 2001 used as covariates in the statistical analysis as such simulations were based on recent historical weather/climate conditions. Our module calls the initiation model five times for each time step and uses the sum of predicted counts for each landscape. Each call uses a different year of weather data. Otherwise, the data analyzed for our statistical model was consistent with the operational scale of Tardis and no scaling was necessary. We assessed the validity of the new initiation module by comparing the number of initiations predicted in the first five years of the simulation to five years of observed historical data corresponding to the years of weather data used in the simulation.

Each fire that is initiated has a Bernoulli probability of exceeding 3 ha – considered and escape - estimated from empirical data (Cumming 2005). An escaped initiation represents a fire that was not suppressed through initial attack by fire operation crews. The final size of escaped fires is sampled from a truncated exponential distribution (Cumming 2001b) that includes the shape parameter, σ_i specified by the composition of the landscape where the fire was initiated. The parameters of the shape parameter were estimated from regional empirical data and result in smaller fires simulated in recently disturbed areas; these include areas harvested and burnt within the last 20 years. The composition of each fire is defined by a multivariate function of forest composition in each affected landscape (Cumming 2001a) and the landscape composition of its neighbours, where neighbours are defined according to the size of the fire. We ran two scenarios in Tardis, one with harvesting and fire, and one where fire was the only disturbance process in the region.

We measured two indices to describe the regional effects of a mixed disturbance regime (harvest and fire) on the system. The first, ΔYEAR , was the number of years between two consecutive disturbances in a given stand. We tracked the frequency of disturbance (BURNED or HARVESTED) for every stand over the 100 years of simulation. Within the disturbance history HARVESTED – BURNED (H – B) was a stand that was harvested then burned,

BURNED – BURNED (B – B) was a stand that was burned and then burned again. We compared the distribution of ΔYEAR between H – B and B – B in the harvested scenarios, and with B – B in the un-harvested simulations. The second index was the cumulative area affected by each combination of disturbance types over the 100 years of simulation.

Results

The ZINB model was the most parsimonious model of lightning fire initiation with covariates including: SSR, B_{ALL} , H_{ALL} , BS, BS^2 , MC, MD, and FFMC_{JWL} (Tables 3-4 and 3-5). The SSR was the only variable maintained in the binomial component of the model. The reduction in AIC with the inclusion of SSR was large ($\Delta\text{AIC } 87$), suggesting should be maintained in the model despite an effective sample size of 8 years, rather than 4704 voxels. The $\text{Pr}(Y = 0|\text{SSR})$ ranged from 0.017 to 0.63. The expected number of initiations decreased with the area burned within 30 years (B_{ALL} ; Table 3-5). In contrast, initiation increased with the area harvested within 30 years (H_{ALL} ; Figure 3-1). The relative effect size for a landscape composed of 10% H_{ALL} was 1.5 (i.e., $e^{0.1 \times 4.06}$), and for B_{ALL} was 0.9 (i.e., $e^{0.1 \times -1.24}$). There was insubstantial interaction between year, H_{ALL} or B_{ALL} , all p -values for interaction terms ranged between 0.1 and 0.8, except for the interaction between B_{ALL} and the year 1995, where the interaction was significant with a p -value of 0.02. A conditioning plot of initiation as a function of B_{ALL} conditioned by year suggested the effect was not strong relative to the remaining, insignificant interactions.

To represent the changes in effect size, we plotted the expected initiation for landscapes when the featured area was replaced by either burned or harvested (B_{ALL} or H_{ALL}) ranging between 0 to 4 000 ha, using parameter estimates from the best model (Table 3-5). The extent of the abscissa was selected to correspond to the range of H_{ALL} observed in the data (Table 3-4). Figure 3-1 illustrates multiple scenarios of the expected change in initiation per voxel. The expected value is presented on the ordinate axis. In the left panel the composition of a landscape is altered via the abscissa as MC, MD, BS, and B_{ALL} are systematically replaced by

H_{ALL} . In the right panel the composition of a landscape is altered along the abscissa as MC, MD, BS, and H_{ALL} are systematically replaced by B_{ALL} . Landscape composition was varied for each scenario but the baseline was set to the mean observed value of each covariate. For example, as the area of H_{ALL} (Figure 3-1 left panel) or B_{ALL} (Figure 3-1, right panel) increased, the area of MC, MD, BS, B_{ALL}/H_{ALL} (respectively) decreased. Weather covariates were set to their mean observed values and did not vary among scenarios. Prediction bands around the ZINB expectations were calculated using 1000 Monte Carlo simulations of standard errors from parameters estimated in the best model. Harvesting activity dominantly focuses on deciduous species (MD) in this region, and secondarily on white spruce conifer (MC). In MD and MC (mesic) forests, harvesting results in a transition of forest cover from a negative relationship with initiation (MD) to a positive one (H_{ALL}), and an increase in initiation from MC to H_{ALL} . In contrast, burning (B_{ALL}) resulted in a negative relationship with initiation. Fires burn more white spruce (MC) and black spruce (BS) than deciduous (MD) relative to their proportional occurrence on the landscape (Cumming 2001a); large areas of BS are transformed to B_{ALL} , and only small areas of MD are affected.

The variance of residuals was higher in 1998 and 1999 than in other sample years. These two years had the greatest number of initiations, JWL days and SSR. We suggest that an even better statistical description of initiations in these 'extreme' fire years might require additional covariates representing more subtle, or interacting biophysical conditions. The goodness of fit calculated using a parametric bootstrap of SS indicated the best ZINB model sufficiently described the distribution of the data ($SS = 1143$, $p = 0.42$, $n = 1000$). There was limited difference between the observed and predicted probabilities of counts generated by the best model. The model fit most poorly for voxels with a single initiation, with a difference between observed and predicted probabilities of 0.032. There was no difference in the significance or selection of parameters using the smaller dataset with non-backdated data so we concluded that our back-dating methods did not influence the results of statistical analyses.

Fine-scale evaluation of statistical patterns

The positive relationship between the number of initiations and amount of harvested forest in a landscape estimated by the previous statistical model implies that fires are more likely to start in harvested stands, compared to other vegetation types or than random. We tested this inference at a finer spatial scale using marked point data in a subset of 171 voxels where an area of 1 000 ha or more had been harvested. Fire operations staff members estimate spatial locations and cover types of initiations during their assessment and fire fighting. There were 73 initiations recorded in the sample voxels. Since these details were not always precisely collected, we calculated a 100 m radius buffer around each recorded location and classified the AVI to indicate all areas of the H_{ALL} cover type. Twenty eight of the 73 fires (38%) were recorded as initiating within the H_{ALL} cover type. If initiation was a spatially random process we would expect nine fire initiations in the harvested cover type given the area of H_{ALL} in the selected region. The values are significantly different using an exact binomial test ($p = 4.7 \times 10^{-8}$). Our fine scaled assessment supports our regional statistical analysis that the probability of initiation increases as a function of the area of recently harvested stands.

Simulation results

Stand level disturbance (by harvesting or burning) over 100 years varied between zero and four events. For those stands disturbed by two or more events, 78% of stands were disturbed twice, 19% were disturbed three times and 3% were disturbed four times. For stands that experienced more than two disturbances we divided the disturbance history into pairs. For example, if a stand was burned in 2010, harvested in 2080 and burned in 2085 it had two disturbance pairs: 2010:2080 and 2080:2085. The mean $\Delta YEAR$ between pairs in the harvested scenario was 26.3 for H - B and was significantly different from 37.4 for B - B ($t = -167$, $df = 222\ 077$, $p = 2.2 \times 10^{-16}$), the mean interval in the unharvested scenario was 37.5 for B - B (Figure 3-3). The lower inter-quartile range was

between five and ten for H – B but between 20 and 30 for B – B. The minimum Δ YEAR between pairs for H – B was zero, and for B – B combinations was five. Tardis allocates harvest decisions prior to fire within a given time step, allowing compound disturbance to initiate within a time step. Based on the order of operations within Tardis it is impossible to incur multiple burns on a stand within a five year step, imitating behaviour observed in nature.

The mean area (over 10 simulations and 100 years) affected by H – B was 247 294 ha, corresponding to an area of 24 729 ha/decade - or 2.5 landscapes per decade. Approximately 25% of the study region is harvestable mesic forest such that 1% of that area is affected by H – B events. The mean area affected by B – B was 639 165 ha in the harvested and 734 468 ha in the un-harvested scenario, corresponding to an area of six and seven landscapes per decade, respectively. Forest harvest occurs on MC and MD stands, whereas fires burn proportionally more BS, JP and MC than MD. Accordingly, the H – B events occur at a reduced interval, and are affecting different areas than the B – B events.

Discussion

This study quantifies the effect of forest management on lightning fire initiation in the mixedwood boreal forest over broad spatial and temporal scales. We found that recently (< 30 years) harvested and burned stands differ in their effect on the lightning fire disturbance regime; the probability of initiation increases with the area of recently harvested forest and decreases with the area of recently burned forest. The harvest effect was detected for areas burned within 30 years but a strong component of the effect came from aspen stands up to 10 years post harvest, suggesting that finer scaled studies might focus on quantifying inter-decadal variation in fire hazard. The true duration of the harvest effect is unclear since our sampling protocol categorized stands as either harvested or burned for 30 years, after which time they were classified according to stand cover attributes. Our fine-scaled validation confirmed that initiation is more likely to occur within recently harvested stands than expected by random. The data for this study was collected over a relatively short period of 8 years however there was no

significant interaction between year and terms for the area burned or area harvested suggesting that the estimates are systematic and meaningful. The emulation of natural disturbance (fire) through harvesting is not supported for the process of subsequent fire initiation. We evaluated the consequences of these opposing relationships by integrating our statistical model of initiation into a regional simulation model, Tardis. The simulation illustrated that the positive relationship between harvest and initiation manifested as a reduction in the disturbance interval for harvested stands (HARVESTED – BURNED), resulting in an accelerated, compound disturbance that is more frequent than the natural disturbance interval (BURNED – BURNED) *sensu* Paine et al. (1998). The outcome is an alteration of the fire regime managers may seek to emulate. It is difficult to determine what disturbance regime we should emulate with forest management. Historical disturbance regimes that produced the observed forest are a temporal accumulation of multi-scale interactions and feedbacks among weather, topography, fuel variation and human intervention. As a result, current forest characteristics may not be appropriate benchmarks for the future given that recent historical data are sampled from a non-equilibrium system (Johnstone and Chapin 2003), and changes in climate are expected to affect biophysical processes in the future (IPCC 2001). We argue that given the relative intactness of this study system, statistical and observational signals in recent historical data provide some indication of the disturbance regime the ecosystem is most closely adapting with.

The ignition of a lightning fire requires fine fuels for initial combustion. We found that harvesting increases initiation rates for at least three decades though our ten year age classes may smooth variation in fuel characteristic. Standing harvest residuals may act as conduits for lightning to strike within a harvested stand where dry and abundant fine fuels (Kiil 1968; Song 2002; Baxter 2004) then combust and spread the fire so that it becomes large enough to be detected. In a synthesis of harvest-fire observations for this region, Baxter (2004) suggested there is a difference in the flammability of conifer and deciduous harvest slash for up to two years post-harvest due to the fine fuels such as needles and young branches, and that by eight years they do not differ. This is consistent

with our treatment of deciduous and coniferous harvest as a single category of fuel/cover type. The fire hazard associated with fine fuels from harvesting slash is considered transitory, decreasing with increasing decay and disintegration (Kiil 1968). Within a few years, areas that contained aspen prior to harvesting will typically regenerate through a period of lush aspen growth from asexual recruitment of 'suckering' shoots (Song 2002); these sites represent most of the area harvested in the study region. Aspen stands rapidly develop towards maximum leaf area and canopy density during 10 – 25 years after a fire. This lush, deciduous growth is thought to reduce the potential for fire spread (Forestry Canada Fire Danger Group 1992). However Greenway et al. (2002; in Song 2002) suggest aspen regeneration and canopy development may be inhibited in post-harvest stands, as compared to post-burn stands, due in part to competition between young aspen and grasses such as *Calamagrostis canadensis* (Michx.) in harvested stands (Hogg and Lieffers 1991). Differences in fire severity have substantial influence on regeneration of boreal tree communities (Johnstone and Chapin 2006); it is not surprising that regeneration would vary between disturbance types. *Calamagrostis* is a common grass in the boreal that colonizes open areas (Greenway and Lieffers 1997) and can survive in the understory as mixedwood stands grow back after disturbance, but tends to die off when light levels below fall below 10% of above canopy light (Greenway and Lieffers 1997). In stands where *Calamagrostis* was present prior to harvesting, it may quickly take over a harvested site and compete with aspen (Greenway et al 2002; in Song 2002). Persistent, standing dead *Calamagrostis* shoots and litter may result in increased fire hazard (Forestry Canada Fire Danger Group 1992). Thus abundant slash material and fine fuel provided by grasses might remain readily combustible and available for fire initiation despite the regeneration of young deciduous trees. Hogg and Lieffers (1991) suggest that *Calamagrostis* dominates after low severity burns or harvesting whereas fireweed (*Epilobium angustifolium* L.) is favoured after severe fire and has reduced fuel hazard due to rapid disintegration of dead shoot material.

The structure of the remaining, standing forest may also contribute to the patterns of initiation we observed. Forest fragmentation associated with harvest activity might result in increased drying at forest edges (Ray et al 2005; Lee 2002 in Song 2002) such that ecotones are more likely to ignite and persist to become detected initiations. Landscape structure has been demonstrated to influence meso-climatology (Pielke and Avissar 2000) such that breezes or even convective storms might be generated over harvested landscapes differently than over those that are burned. Standing snags and patches that remain after a fire would reduce surface air flow across burned areas whereas increased air flow across harvested stands could result in accelerated dessication of fine fuels that are involved in combustion and propagation of a lightning fire.

Post-harvest burning is used as a treatment to reduce subsequent fire hazard and improved regeneration in other regions. Stephens (1998) demonstrated variation in fire behaviour through a suite of harvest/biomass removal/burn treatments in Californian mixed-conifer forests, showing that post-harvest burn treatments always resulted in the reduction of subsequent fire behaviour. While forest managers have suggested that it might be necessary to burn harvest blocks in their biodiversity management plans for mixedwood boreal forests, this treatment might also be prudent to reduce subsequent fire initiations. However the perceived risk of hold-over or escaped fires precludes its systematic use in this study region (Baxter 2004). From economic, management and conservation perspectives, the probability of fire escape from slash burning prescriptions in contrast to the probability of subsequent lightning fire initiation, and the benefit of fire to the whole ecosystem should be calculated to determine where and when fires should be set or let to burn.

Variation in initiation probability among landscapes was related to the relative composition of black spruce, mature aspen and white spruce, in addition to harvested and burned areas. Initiation increased in landscapes containing a greater area of spruce (black and white) and decreased in those containing a greater area of aspen. Our findings are consistent with previous research in the region that demonstrated evidence of a biotic and abiotic regulation of initiation

occurrence by forest cover type and weather for a time series prior to harvesting activity (Krawchuk et al. 2006). A lightning fire can not occur without appropriate weather conditions; annualized indices of daily fire weather and lightning (FFMC_{JWL}), and regional weather (SSR) showed that initiation was also strongly regulated by weather conditions. The weather variables explained slightly more variation of initiation counts than in the previous analysis of initiation occurrence (Krawchuk et al. 2006).

The simulation experiment showed that the increased frequency of initiations in harvested stands reduced the interval between consecutive disturbances for previously harvested stands when compared to previously burned stands. The majority of harvest occurs in mesic forests containing aspen and white spruce communities so these types were disproportionately affected in comparison to those incurring consecutive burns. Fire rarely burns aspen stands (Quintilio et al. 1991; Van Wagner 1977), so the short disturbance interval introduced by the harvest-burn compound effect is novel to the system. The Δ YEAR index does not explicitly integrate area, but indicates that many harvested stands were re-disturbed (burned) before the expected next harvest and, in the case of white spruce, before mature, cone-bearing individuals could grow to contribute seed to the area. Stand composition is influenced by the interaction of species' life history strategies with the frequency and type of disturbance in and around the stand (Johnstone and Chapin 2006a; Weir et al. 2000). Johnstone and Chapin (2006b) illustrated that shorter fire return intervals (< 30 years) resulted in decreased densities of conifer regeneration relative to expected pre-fire community dominance by conifers in the boreal forest of British Columbia and the Yukon, Canada. Fire return interval did not influence aspen stem densities (Johnstone and Chapin 2006b), but post-harvest competition with grasses could reduce vegetative aspen regeneration. Weir et al. (2000) suggested that increased fire frequency in Saskatchewan's mixedwood boreal forests resulting from nearby homesteading reduced or locally exterminated species with older ages of maturity, no serotinous cones, and no vegetative reproduction, such as white spruce. A decrease in conifer recruitment through accelerated disturbance could add even

more pressure to concerns of 'unmixing' the mixedwood boreal (Cumming 2001a; Weir et al. 2000) due to a reduction in white spruce populations. The cumulative area affected by the overlapping disturbance of harvest and burn was approximately 1% of the merchantable forest, amounting to a local peculiarity rather than a broad scale effect. However increases in fire and harvest activity expected with climate change and development could expand the area affected by compound disturbances in the future. Since our study region is a managed forest, silvicultural intervention such as planting or site preparation could mediate some of these outcomes but there is currently no stipulation to re-stock harvested areas that are subsequently burned.

We presented 10 independent simulations of each scenario (harvested and un-harvested) and the outcome of this comparison depends on the integration of many decision rules coded within Tardis. The current software does not include a relationship between forest harvest, fire escape and fire size. If these effects are strong, they could increase the area of compound disturbance initiated by forest harvest. We have described and implemented simple changes to Tardis to isolate the effects of the burn and harvest parameters on the fire initiation process and explore what this might mean for forest dynamics. Tardis is a calibrated representation of both the harvest and ecological dynamics of the region but it does not include dynamics or footprint of oil and gas development (Schneider et al. 2003) or human-caused fire (Arienti et al. 2006) so cannot be thought to represent the current state of the region.

Paine et al. (1998) described compound disturbance as 'disturbances that have occurred either simultaneously or in a sequence rapid enough that recovery from the single pulse has not significantly progressed'. Compound disturbances are suggested to initiate transition among alternative stable ecosystem states (Paine et al. 1998; Jasinski and Payette 2005), but Paine et al. (1998) posit that three or more pulses may be required to initiate this change. This can include multiple disturbances of the same or different kind, occurring naturally or induced anthropogenically. In our study region salvage harvest could also contribute to compound disturbance (Lindenmayer et al. 2005; Donato et al. 2006), but this

factor was not included in our models. The amount of disturbance a system can experience without a change in state via variables and processes that control behaviour is referred to as its ecological resilience (Holling 1973). Compound disturbances by harvest, insects and fire have resulted in short-term alteration of spruce forest communities in northern Quebec (Payette and Delwaide 2005). Insect outbreaks followed closely by fire activity are suggested to be responsible for the transition between spruce-moss forest and spruce-lichen woodland states in the same region (Jasinski and Payette 2005). Though an "alternative stable state" (Holling 1973; Scheffer et al. 2001; Beisner et al. 2003) has not yet been identified in the mixedwood boreal forest, a *Calamagrostis*-dominated community assemblage (introduced above) could be considered a candidate. Fires burning through recently harvested areas, carried by grasses and fine fuels may be less intense than those burning abundant or larger fuels in stands with mature trees conducive to crowning. These low-intensity fires may perpetuate a feedback and recruitment of grasses (Hogg and Lieffers 1991; Brooks 2004) suggestive of an alternative stable state under environmental conditions also suitable for mixedwood forest growth. In our simulation experiment the majority of disturbed stands incurred only two pulses over 100 years, but three or four pulses also occurred, even without the inclusion of a grass-invasion cover type to act as a vegetation feedback to disturbance. As these compound disturbances accumulate, a change in state favouring *Calamagrostis* could be further supported through ongoing changes in climate that are predicted to continue across the western mixedwood boreal region (Hogg and Bernier 2005).

Table 3-1. Observed area burned (ha) by lightning and human-caused fires, and area harvested (ha) in the study region from 1994-2001 calculated from the Alberta Vegetation Inventory.

Year	Burned	Harvested
1994	1 283	8 925
1995	68 991	13 254
1996	318	5 223
1997	13	10 662
1998	37 195	9 057
1999	553	5 126
2000	558	7 937
2001	1 173	2 421
Total	110 084	62 605
Mean	13 760	7 826

Table 3-2. Summary of initiation data (per voxel). Overdispersion of the count data appears to be driven by specific years of high counts.

Year	Initiations	Mean	Variance
1994	82	0.14	0.15
1995	77	0.13	0.15
1996	15	0.026	0.025
1997	39	0.066	0.062
1998	212	0.36	0.42
1999	289	0.49	0.62
2000	85	0.14	0.16
2001	67	0.11	0.12
1994-2001	866	0.18	0.24

Table 3-3. Summary of joint weather-lightning indices (JWLs) for Fine Fuel moisture code (FFMC) and Duff Moisture Code (DMC), and the annual, regional seasonal severity rating (SSR).

Year	FFMC _{JWL} Mean/voxel	FFMC _{JWL} Annual total	DMC _{JWL} Mean/voxel	DMC _{JWL} Annual total	SSR Annual
1994	2.3	1 358	2.4	1 399	1.8
1995	1.1	645	2.4	1 428	2.5
1996	0.7	425	0.1	77	0.7
1997	2.1	1 237	0.1	61	0.8
1998	4.5	2 643	7.6	4 495	3.3
1999	3.3	1 910	1.4	807	2.0
2000	1.0	581	0.0	12	0.7
2001	2.0	1 200	0.9	534	1.8

Table 3-4. Summary of the 23 cover types classified from the Alberta Vegetation Inventory polygons. We sampled 588 landscapes with a mean area of 9448 ha.

³Values are presented for 2001. The * and @ indicate variables collapsed to B_{ALL} and H_{ALL}, respectively, for regression analysis (see *Exploratory data analyses*).

Category description	Moniker	Range (ha/voxel)	³ Percent (region)	³ Incidence (# of landscapes)
Deciduous dominated mesic, 90+ years	MD ₉₀₊	0-3397	4%	550
Deciduous dominated mesic, 31 to 90 years	MD	0-7174	16%	587
Conifer dominated mesic, 31+ years (white spruce)	MC	0-2914	5%	580
Black spruce (including larch and some birch/black spruce mixes), 31+ years	BS	161-8762	44%	588
Jack pine (including deciduous dominated stands where pine is the dominant conifer), 31+ years	JP	0-6781	8%	580
Open (including wetlands and non-vegetated, natural areas)	O	36-5241	8%	588
Water	WA	0-3298	5.1%	551
Burns, 1 to 10 years*	B ₁₀	0-7129	2%	77
Burns, 11 to 20 years*	B ₂₀	0-7718	4%	148
Burns, 21 to 30 years*	B ₃₀	0-7156	1%	112
All burns, 1 to 30 years	B _{ALL}	0-8256	7%	240

continued

Burns, 0 up to 1 year (current year)	B ₀	0-7125	0.03%	3
Burns of unknown origin	B _U	0-707	0.1%	157
Conifer harvest, 1 to 10 years [@]	HC ₁₀	0-1951	0.2%	93
Conifer harvest, 11 to 20 years [@]	HC ₂₀	0-2003	0.2%	61
Conifer harvest, 21 to 30 years [@]	HC ₃₀	0-760	0.1%	46
Aspen harvest, 1 to 10 years [@]	HD ₁₀	0-1667	0.9%	196
Aspen harvest, 11 to 20 years [@]	HD ₂₀	0-1219	0.1%	74
Aspen harvest, 21 to 30 years [@]	HD ₃₀	0-1260	0.3%	114
All harvest, 1 to 30 years	H _{ALL}	0-3636	1.8%	264
Salvage harvest, 1 to 30 years	H _{SAL}	0-788	0.08%	31
Conifer and aspen harvest, 0 up to 1 year (current year)	H ₀	0-1503	0.05%	24
Harvest of unknown origin	H _U	0-987	0.4%	191
Anthropogenic features (including well-sites, seismic lines, clearings, roads)	ANTH	0-2540	0.8%	508

Table 3-5. Parameters and standard errors estimated by the best ZINB model (log of the likelihood= -2246.19). Calculated *p*-values are presented from the Wald test statistic. A change in the AIC with the removal of each term is indicated by Δ AIC. The binomial component of the model estimated the $\text{Pr}(Y=0)$ so the $\text{Pr}(Y=1)$ is the opposite sign thus there is a positive relationship between fire initiation per voxel and annual SSR.

Parameter	Estimate	Standard Error	P-value	Δ AIC
Intercept (B)	+1.78	0.28	3.78×10^{-10}	
SSR	-1.76	0.35	5.54×10^{-7}	87
Intercept (NB)	-1.94	0.32	1.29×10^{-9}	
H _{ALL}	+4.06	1.13	3.30×10^{-4}	10
B _{ALL}	-1.24	0.45	5.58×10^{-3}	6
MC	+1.87	0.86	2.91×10^{-2}	3
MD	-2.31	0.47	1.06×10^{-6}	22
BS	+2.45	1.029	1.70×10^{-2}	4
BS ²	-2.50	1.026	1.92×10^{-2}	4
FFMC _{JWL}	+0.13	0.018	4.93×10^{-13}	50
Log(Theta)	0.85	0.33	1.033×10^{-2}	10

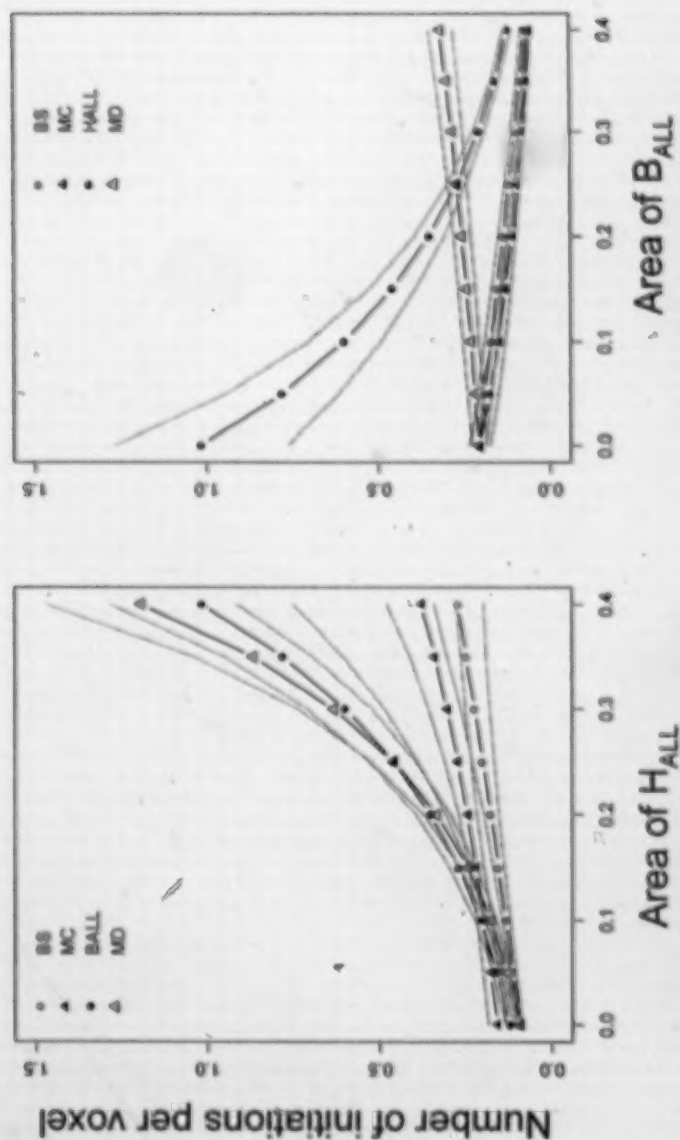
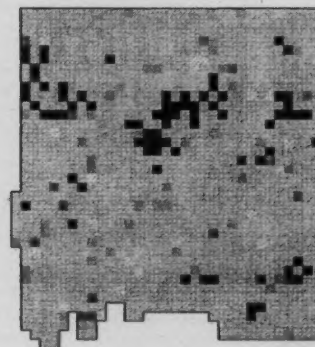


Figure 3-1. Changes in expected fire initiation per voxel associated with changing landscape composition, prediction bands are illustrated in grey. Based on the scenario described in the text, the expected number of initiations increases as black spruce (BS), white spruce (MC), burned forest (B_{ALL}) and aspen forest (MD) are replaced by harvested forest (H_{ALL}) (left). The expected number of initiations decreases as MC, BS and H_{ALL} are replaced by B_{ALL} , and increases as MD is replaced by B_{ALL} (right). There is a large effect on predicted initiation associated with the replacement of B_{ALL} by H_{ALL} , illustrated by '••' in both panels.

Area burned



0 : 23
24 : 38
39 : 50
50 : 65
66 : 82
83 : 114
115 : 217
218 : 367
368 : 1110
1111 : 2350
2351 : 5410
5411 : 8334
8335 : 9321
9322 +

Figure 3-2. A map of the study region generated from the Tardis model illustrating the simulated spatial pattern of area burned in a single time step. Each square in the map is a landscape with an area of $\approx 10\,000$ ha; stand attributes are collated and updated within each landscape using an aspatial stand list.

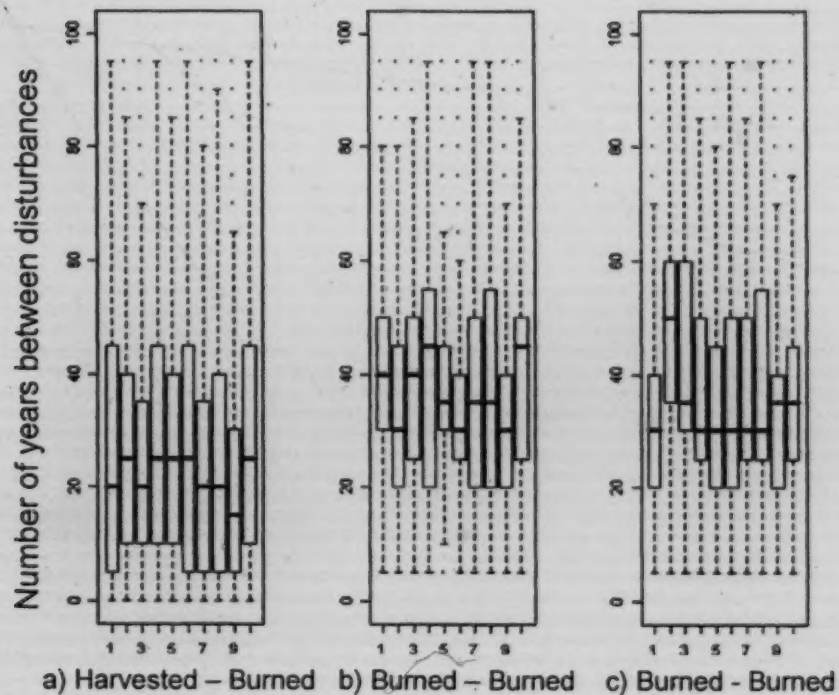


Figure 3-3. The boxplot distribution of the return interval between two consecutive disturbances for stands in harvested (a & b) and un-harvested (c) scenarios as shown by box and whisker plots from 10 simulations. Harvested - Burned refers to areas where a harvesting followed by fire, Burned - Burned refers to areas burned repeatedly.

Literature Citations

- Albani, M., D.W. Andison, and J.P. Kimmins. 2005. Boreal mixedwood species composition in relationship to topography and white spruce seed dispersal constraint. *Forest Ecology and Management* **209**: 167-180.
- Alberta-Pacific Forest Industries, Inc. 2004. Alberta-Pacific FMA Area Forest Management Plan September 2004. Boyle, Alberta, Canada.
- Alberta Sustainable Resource Development [ASRD]. 2004. Ten year statistical summary: Human and lightning caused wildfires. http://www.srd.gov.ab.ca/wildfires/fpd/wi_hdtys_10_yr_average.cfm; accessed 9 June, 2006.
- Alberta Sustainable Resources and Development [ASRD]. 2006. Historical Wildfire Database. http://www.srd.gov.ab.ca/wildfires/fpd/wi_hdhwd.cfm; accessed 9 June, 2006.
- Alexander, M.E. 1982. Fire behavior in aspen slash fuels as related to the Canadian Fire Weather Index. *Canadian Journal of Forest Research* **12**:1028-1029.
- Anderson, K., and P. Englefield. 2001. Quantile characteristics of forest fires in Saskatchewan. Pages 9-16 in *Proceedings of the Fourth Symposium on Fire and Forest Meteorology*, 13-15 November, 2001, Reno Nevada. American Meteorological Society, Boston, Massachusetts.
- Arienti, M.C., S.G. Cumming and S. Boutin. 2006. Empirical models of forest fire initial attack success probabilities: the effects of fuels, anthropogenic linear features, fire weather, and management. *Canadian Journal of Forest Research* **36**:3155-3166.
- Armstrong, G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. *Canadian Journal of Forest Research* **29**:424-433.
- Armstrong, G.W., W.L. Adamowicz, J.A. Beck, S.G. Cumming, and F.K.A. Schmiegelow. 2003. Coarse filter ecosystem management in a nonequilibrating forest. *Forest Science* **49**:209-223.
- Baxter, G. 2004. Managing of harvest debris in east-central Alberta. *FERIC, Advantage* **5**:1-10.
- Beisner, B.E., D.H. Haydon and K. Cuddington. 2003. Alternative view on alternative stable states. *Frontiers in Ecology and the Environment* **1**:376-382.
- Bessie, W.C. and E.A. Johnson. 1995. The relative importance of fuels and

- weather on fire behavior in subalpine forest. *Ecology* **76**:747-762.
- Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS.
<http://www.spatial ecology.com/htools>.
- Bigler, C., D. Kulakowski, and T.T. Veblen. 2005. Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. *Ecology* **86**:3018-3029.
- Chen, H.Y. H. and R.V. Popadiouk. 2002. Dynamics of North American boreal mixedwoods. *Environmental Reviews* **10**:137-166.
- Cumming, S.G. 2001a. Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn? *Ecological Applications* **11**:97-110.
- Cumming, S.G. 2001b. A parametric model of the fire-size distribution. *Canadian Journal of Forest Research* **31**:1297-1303.
- Cumming, S.G. 2005. Effective fire suppression in boreal forests. *Canadian Journal of Forest Research* **35**:772-786.
- Cumming, S.G. and G.W. Armstrong. 2001. Divided land base and overlapping forest tenure in Alberta, Canada: a simulation study exploring costs of forest policy. *Forestry Chronicle* **77**: 501-508.
- Cumming, S.G. and G.W. Armstrong. 2004. Divided landbase, overlapping tenures, and fire risk. *Forestry Chronicle* **80**:478-484.
- Cumming, S.G. and P. Vernier. 2002. Statistical models of landscape pattern metrics, with applications to regional scale dynamic forest simulations. *Landscape Ecology* **17**:433-444.
- Donato, D.C., J.B. Fontaine, J.L. Campbell, W.D. Robinson, J.B. Kauffman and B.E. Law. 2006 Post-wildfire logging hinders regeneration and increases fire risk. *Science* **311**: 352-352.
- Efron, B. and R.J. Tibshirani. 1998. *An Introduction to the Bootstrap*. Chapman and Hall/CRC, New York.
- ESRI Inc. 2005. Redlands, CA.
- Flannigan, M.D. and B.M. Wotton. 1989. A study of interpolation methods for forest fire danger rating in Canada. *Canadian Journal of Forest Research* **19**:1059-1066.
- Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian Forest Fire Behavior Prediction System. Forestry Canada, Ottawa.

- Gillson, L. and K.J. Willis. 2004. 'As earth's testimonies tell': wilderness conservation in a changing world. *Ecology Letters* 7:990-998.
- Greenway, K.J. and V.J. Lieffers. 1997. A boreal forest grass with an open meadow photosynthetic strategy. *Canadian Journal of Botany* 75:562-567.
- Gunderson, L.H., and C.S. Holling. 2002. *Panarchy: Understanding transformations in human and natural systems*. Island Press, Washington, D.C.
- Hély, C., Y. Bergeron, and M.D. Flannigan. 2000. Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. *Journal of Vegetation Science* 11:813-824.
- Hély, C., M.D. Flannigan, Y. Bergeron, and D. McRae. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. *Canadian Journal of Forestry Research* 31:430-441.
- Hirsch, K., V. Kafka, and B. Todd. 2001. Using forest management techniques to alter forest fuels and reduce wildfire size: An exploratory analysis. *in* R.T. Engstrom, and W.J. de Groot, editors. *Proceedings of the 22nd Tall Timbers Fire Ecology Conference: Fire in Temperate, Boreal, and Montane Ecosystems*. Tall Timbers Research Station, Tallahassee, FL
- Hogg, E.H. and V.J. Lieffers. 1991. The impact of *Calamagrostis canadensis* on thermal regimes after logging in northern Alberta. *Canadian Journal of Forest Research* 21:387-394.
- Hogg, E.H.T. and P.Y. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. *Forestry Chronicle* 81:675-682.
- Holling, C. S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4:2-23.
- Hunter, M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. *Biological Conservation* 65:115-120.
- IPCC. 2001. *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, and Johnson CA (eds) Cambridge University Press, Cambridge, United Kingdom
- Jasinski, J.P.P. and S. Payette. 2005. The creation of alternative stable states in the southern boreal forest, Quebec, Canada. *Ecological Monographs* 75:561-583.

- Johnson, E.A., K. Miyanishi and J.M.H. Weir. 1998. Wildfires in the western Canadian boreal forest: Landscape patterns and ecosystem management. *Journal of Vegetation Science* 9:603-610.
- Johnstone, J.F., and F.S. Chapin. 2003. Non-equilibrium succession dynamics indicate continued northern migration of lodgepole pine. *Global Change Biology* 9:1401-1409.
- Johnstone, J.F. and F.S. Chapin. 2006a. Effects of soil burn severity on post-fire tree recruitment in boreal forest. *Ecosystems* 9:14-31.
- Johnstone, J.F. and F.S. Chapin. 2006b. Fire interval effects on successional trajectory in boreal forests of northwest Canada. *Ecosystems* 9:268-277.
- Kiil, A.D. 1968. Changes in the physical characteristics and moisture content of pine and spruce-fir slash during the first five years after logging. Canada Department of Forestry and Rural Development, Forest Research Laboratory Edmonton, Alberta. Internal Report A-14.
- Krawchuk, M.A., S.G. Cumming, M.D. Flannigan, and R.W. Wein. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. *Ecology* 87:458-468.
- Lambert, D. 1992. Zero-inflated Poisson regression, with an application to defects in manufacturing. *Technometrics* 34:1-14.
- Lavoie, N. 2004. Variation in flammability of jack pine/black spruce stands with time since the last fire. Ph.D. dissertation. University of Alberta, Edmonton, Alberta, Canada.
- Lavoie, L. and L. Sirois. 1998. Vegetation changes caused by recent fires in the northern boreal forest of eastern Canada. *Journal of Vegetation Science* 9:483-492.
- Legendre, P. and L. Legendre. 1998. *Numerical ecology*. 2nd English edition. Elsevier Science, Amsterdam, The Netherlands.
- Lindenmayer, D. 2006. Salvage harvesting – past lesson and future issues. *Forestry Chronicle* 82:48-53.
- Martin, T.G., B.A. Wintle, J.R. Rhodes, P.M. Kuhnert, S.A. Field, S.J. Low-Choy, A.J. Tyre, and H.P. Possingham. 2005. Zero tolerance ecology: improving ecological inference by modelling the source of zero observations. *Ecology Letters* 8:1235-1246.
- McIntire, E.J.B., R. Duchesneau and J.P. Kimmins. 2005. Seed and bud legacies interact with varying fire regimes to drive long-term dynamics of boreal forest communities. *Canadian Journal of Forest Research* 35:2765-2773.

- McRae, D.J., L.C. Duchesne, B. Feedman, T.J. Lynham and S. Woodley. 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. *Environmental Reviews* 9: 223-260.
- Nash, C.H. and E.A. Johnson. 1996. Synoptic climatology of lightning-caused forest fires in subalpine and boreal forests. *Canadian Journal of Forest Research* 26:1859-1874.
- Nimchuk, N. 1989. Ground truthing of LLP lightning data in Alberta. Pages 33-40 in D.C. MacIver, H. Auld, and R. Whitwood, editors. *Proceedings of the 10th Conference on Fire and Forest Meteorology*, Ottawa, Ontario. AES, Downsview, Ontario, Canada
- Paine, R.T., M.J. Tegner and E.A. Johnson. 1998. Compounded perturbations yield ecological surprises. *Ecosystems* 1:535-545.
- Payette, S. and A. Delwaide. 2003. Shift of conifer boreal forest to lichen-heath parkland caused by successive stand disturbances. *Ecosystems* 6:540-550.
- Pielke, R.A. and R. Avissar. 1990. Influence of landscape structure on local and regional climate. *Landscape Ecology* 4: 133-155.
- Quintilio, D., M.E. Alexander and R.L. Ponto. 1991. Spring fires in a semi-mature trembling aspen stand in central Alberta. *Forestry Canada Information Report NOR-X-323*. Forestry Canada, Northwest Region.
- R Development Core Team. 2005. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>
- Ray, D., D. Nepstad and P. Moutinho. 2005. Micrometeorological and canopy controls of fire susceptibility in a forested Amazon landscape. *Ecological Applications* 15: 1664-1678.
- Reed, W.J. 2001. Statistical inference for historical fire frequency using the spatial mosaic. In: Johnson, E.A. and K. Miyanishi, *Forest fires: behaviour and ecological effects*. Academic Press. New York.
- Renkin, R.A. and D.G. Despain. 1992. Fuel moisture, forest type, and lightning-caused fire in Yellowstone National Park. *Canadian Journal of Forest Research* 22:37-45.
- Rowe, J.S. 1972. *Forest regions of Canada*. Publication Number 1300. Canadian Forest Service, Ottawa, Ontario, Canada.
- Scheffer, M., S. Carpenter, S. J.A. Foley, C. Folke and B. Walker. 2001. Catastrophic shifts in ecosystems. *Nature* 413:591-596.

- Schimmel, J. and A. Granstrom. 1997. Fuel succession and fire behavior in the Swedish boreal forest. *Canadian Journal of Forest Research* **27**:1207-1216.
- Schneider, R.R., J.B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the Western Canadian Sedimentary Basin: A modelling approach. *Ecology and Society* **7**(1):8.[online].
URL:<http://www.ecologyandsociety.org/vol7/iss1/art8>
- Schoennagel, T., T.T. Veblen, and W.H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. *Bioscience* **54**:661-676.
- SmartWood Program. 2005. Forest management public summary for Alberta Pacific Forest Industries, Inc. Rainforest Alliance, New York. 29 pp.
- Song, S.J. (editor) 2002. Ecological basis for stand management: A synthesis of ecological responses to wildfire and harvesting. Alberta Research Council Inc., Vegreville, AB.
- Stephens, S.L. 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests. *Forest Ecology and Management* **105**:21-35.
- Strong, W.L. and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands, and Wildlife, Land Information Division, Resource Information Branch, Edmonton, Alberta.
- Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* **7**:23-34.
- Van Wagner, C.E. 1987. Development and structure of the Canadian Forest Fire Weather Index System, Forestry Technical Report 35. Canadian Forestry Service, Ottawa, Canada.
- Weir, J.M.H., E.A. Johnson and K. Miyanishi. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. *Ecological Applications* **10**:1162-1177.

Chapter 4. Predicted changes in fire weather suggest increases in lightning fire initiation and future area burned in the mixedwood boreal forest*

Introduction

Natural disturbances are regulated by the interplay of biotic and abiotic factors (Paine and Levin 1981; Pickett and White 1985). Forest fire is the dominant natural disturbance in Canada's western boreal forest (Stocks et al. 2002) where the fire regime of a given area reflects the integration of weather conditions, ignition agents, fuel variability, and fire suppression (Schoennagel et al. 2004). The ongoing feedback between fire, weather and vegetation contributes to the community of species that persist in the system (Greene and Johnson 1999; Johnson et al. 2001). Global climate change is predicted to alter weather patterns from those observed historically (IPCC 2001) and fire activity is expected to reflect these changes (Overpeck et al. 1990). Shifts in vegetation communities and species ranges resulting directly from successful migration of propagules in response to changing bioclimatic envelopes are expected, but will occur at a relatively slow rate (Weber and Flannigan 1997; Clark 1998; Johnstone and Chapin 2003). Changes in weather may influence community structure more rapidly through fire-mediated transformation of vegetation where the post-disturbance establishment of species is influenced by weather conditions (e.g., Hogg and Wein 2005), and/or where increases in disturbance frequency alters community structure (Paine et al. 1998; Jasinski and Payette 2005; McIntire et al. 2005; Johnstone and Chapin 2006). Forecasting changes in fire activity as a function of climate change is one step towards understanding the future dynamics of this boreal ecosystem.

Many methods can be used to study the relationship between climate change and fire but climate change simulation models provide choice weather data for studies looking forward in time. In Canada, fire weather conditions are evaluated using the Fire Weather Index (FWI) System developed by the Canadian

* A version of this chapter has been submitted for publication. Krawchuk et al. In review. *Climatic Change*.

Forest Service (Van Wagner 1987). The FWI system integrates daily precipitation, humidity, temperature and wind speed into a suite of indices such as the Fine Fuel Moisture Index or the Seasonal Severity Rating. There is some discrepancy of results from analyses using data from climate change models to predict future fire activity in eastern Canada (Flannigan et al. 1998; Flannigan and Van Wagner 1991; Flannigan et al. 2005) however results generally agree that future weather conditions will contribute to more extreme calculated FWI indices in the western boreal of Canada (Flannigan et al. 1998; Li et al. 2000; Parisien et al. 2005), along with a longer fire season (Wotton and Flannigan 1993) and an increase in area burned (Flannigan et al. 2005). These patterns have a host of contributing processes which may interact in non-linear ways. For example an increase in area burned could result from more fire initiations, larger fires, or more and larger fires. We suggest that by breaking fire activity into phases and assessing the influence of weather conditions on each phase individually, a more informative, hierarchical estimate of future fire activity can be derived.

Lightning fires produce the majority of area burned in Canada (Weber and Stocks 1998). However predictions of spatio-temporal patterns in lightning activity from Global Climate Model (GCM) or Regional Climate Model (RCM) data in Canada are not currently available, though this genre of research is underway (J-P Blanchet, Université du Québec à Montréal, pers. comm.). In previous work we illustrated a positive relationship between empirical patterns of historical (1983 – 2001) lightning fire initiation and two measures of fire weather over a large study region in the boreal mixedwood forest (Krawchuk et al. 2006; Krawchuk and Cumming in preparation). We used a joint fire weather lightning index to integrate simultaneous instances of low fuel moisture and lightning activity, and the seasonal severity rating to quantify annual variation in lightning fire risk independent of lightning activity. Though lightning is (obviously) a fundamental element of lightning fire initiation, the latter index illustrates that inferences about patterns of lightning fire initiation can be made based on weather characteristics independent of explicit information on lightning activity.

This study had three objectives. The first objective was to develop a predictive model of lightning fire initiation based on empirical data quantifying the relationship between initiation and a suite of weather indices, independent of lightning activity. We developed five annual indices based on the Daily Severity Rating (DSR) of the FWI System (Van Wagner 1987) and selected those variables that resulted in the most parsimonious model of annual counts of lightning fire initiation for 588 landscapes in the mixedwood boreal forest in central-eastern Alberta, Canada from data collected between 1983 and 2001. The second objective was to use this parametric model to predict future rates of lightning fire initiation based on weather data provided by a regional simulation of climate change. We used parameter estimates from objective one to predict lightning fire initiation under $1 \times \text{CO}_2$ (1975-1985), $2 \times \text{CO}_2$ (2040-2049) and $3 \times \text{CO}_2$ (2080-2089) FWI conditions interpolated from the Canadian RCM (CRCM; Laprise et al. 1998). By necessity, these predictions assume the relationship between lightning activity and weather variables is time and space-invariant. The third objective was to estimate the magnitude of change in area burned that might result from this simple increase in initiation frequency. We generated estimates of the area burned in each year by incorporating our estimates of future lightning fire initiation (objective two) with existing empirical estimates of the number of fire initiations that grow to be large fires (fire escapes; Cumming 2005) and the fire size distribution (Cumming 2001a). These data provided an estimate of future changes in area burned due to changing fuel moisture conditions conducive to lightning fire initiation, independent of additional changes to other phases of fire activity.

Historical patterns of fire weather and lightning fire initiation

Methods

Study area

The study region was a 58 000 km² of boreal mixedwood forest in central-eastern Alberta, Canada. The climate is continental with mean historical winter and summer temperatures of -13.2°C and +13.5°C, respectively and total

precipitation in summer and winter in the range of 240 mm and 64 mm, respectively (Strong and Leggat 1992). Topography is relatively uniform with elevation of approximately 200 to 1000 m. Glacial movement through the region has resulted in rolling morainic deposits on the uplands and smoother glacio-lacustrine deposits on the lowlands (Rowe 1972). Forest tree species are stratified by soil moisture regime (Strong and Leggat 1992). Upland mesic sites are dominated by mixtures of trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.) or white spruce (*Picea glauca* (Moench) Voss). Paper birch (*Betula papyrifera* Marsh) is present but pure stands are rare. White spruce and balsam fir (*Abies balsamea* (L.) Mill) dominate older stands but pure stands are not currently abundant. Xeric sites are vegetated by jack pine (*Pinus banksiana* Lamb), sometimes in association with aspen. Moist, poorly drained sites are dominated by black spruce (*Picea mariana* (Mill.) B.S.P.). There are extensive wetlands including open areas of muskeg bog, fens, swamps and marshes. Tamarack (*Larix laricina* (Du Roi) Koch) is the dominant tree species in fens. Further information about this area can be found in (Krawchuk et al. 2006).

The Province of Alberta is partitioned into townships using an administrative grid system, each township measures approximately 9 600 ha, varying slightly due to geographic corrections. Each township in the study region was considered as a spatial sampling unit, hereafter referred to as a landscape. The study region was divided into three-dimensional space-time samples, called voxels, with the units landscape and year. Data were collected throughout the fire season (May 1 to September 30) from landscapes between 1983 and 2001 and summarized for each year. All spatial analyses were done in ESRI ArcMap 9.1 (ESRI, 2005) using the NAD27 UTM zone 12 projection. All statistical analyses were done in R (R Development Core Team, 2005 version 2.2.0).

Lightning fire initiation and fire weather data

Records of lightning fire initiation were acquired from Alberta Sustainable Resource Development (ASRD 2005). These records described the date, approximate location and cause of all fires detected in Alberta. We registered all

lightning-caused fires detected during the fire season over the study interval to the appropriate voxel. Un-detected or un-recorded fires may have occurred in the region but fire detection efficiency appears to have been constant over the study interval (Cumming 2005). Multiple fires within the same landscape on the same day were a result of a single environmental condition and counted as a single event. If these fires were to spread, they would likely coalesce into a single, larger fire so the ecological implication of synthesizing these data is minimal. We refer to these counts as 'initiations'. Figure 4-1 shows the distribution of the initiation data (mean = 0.17, s.d. = 0.52). There was no evidence for consistent spatial or temporal autocorrelation in counts of initiations per voxel using Bonferroni-corrected local Moran's I coefficients (Krawchuk and Cumming in preparation) or Mantel r-statistics (Krawchuk et al. 2006).

We quantified weather conditions conducive to initiation from empirical data for each voxel using five indices based on the DSR (daily severity rating) of the FWI System. The DSR was calculated from daily data collected from 65 provincial weather stations operated and calibrated by ASRD in and around the region using recorded solar noon values of temperature, relative humidity, wind speed and 24 hour accumulated rainfall (Van Wagner 1987). We interpolated the DSR across the study region to provide estimates of daily conditions for every landscape across the region using an inverse-distance weighting method with six nearest neighbours. This method was validated in the study region by Krawchuk et al. (2006).

We created five annual indices based on the DSR (daily) to represent different hypotheses of weather patterns that might explain the observed spatio-temporal variation in initiation among voxels. These indices were based on the premise that there would be a positive linear or quadratic relationship between a given index and either initiation occurrence or count per voxel since this would represent an increase in the number of days with fire weather appropriate for initiation given a lightning discharge. Indices were assumed to be conditional on lightning activity and can be thought of as providing an appropriate fuel moisture framework within which lightning activity might interact. The indices included

[1] DSR-days, [2] weighted DSR-days, [3] DSR-sequence, [4] DSR-sequence count, and [5] Seasonal Severity Rating (SSR).

[1] DSR-days. We selected each day in the fire season where the interpolated DSR in a landscape was calculated to be greater than three and gave it a value of one. The DSR-days index was the annual sum of these values. The critical value of three was selected based on work by Anderson and Englefield (2001) wherein they quantified the cumulative distribution of observed lightning fire initiations as a function of DSR and showed that 50% of initiations occurred at a DSR of 3.3 or higher. The study done by Anderson and Englefield (2001) was conducted in the boreal mixedwood forest of Saskatchewan, within roughly 500 km of our study region.

[2] weighted DSR-days. We selected each day in the fire season where the DSR in a landscape was greater than three. The value given to each of these days was weighted using the logistic transformation $e^{(DSR-4)}/1+e^{(DSR-4)}$ that resulted in a weighting scheme having the sigmoidal shape of the logistic function that is bounded by zero and unity. This functional form describes the hypothesis of a non-linear increase in initiation associated with increasing values of DSR. Higher levels of DSR approached unity asymptotically under the suggestion that for DSR of roughly nine or greater, the conditional probability of initiation would not increase substantially.

[3] DSR-sequence. We selected each day in the fire season where the DSR in a landscape was greater than three. Each grouping of consecutive days where DSR was greater than three was given a value of $S = n(n+1)/2$, where n is the number of consecutive days, and S is a given sequence. The sum of all S values was calculated for each voxel. This index was based on work by Flannigan and Harrington (1988) that shows a strong relationship between long sequences of 'dry' days and area burned. We hypothesized that voxels with longer sequences of days with a DSR greater than three would be more likely to incur an initiation due to increasing fire danger.

[4] DSR-sequence count. This was a count of the sequences S , calculated by DSR-sequence [3]. Each sequence of days where DSR was greater than three

(S, in DSR-sequence [3]) was given a value of one. The sum of these values was the DSR-sequence count. This variable provided an additional filter to counts calculated by [1] DSR-days; it was a count of the instances when two or more sequential days had a DSR greater than three.

[5] Seasonal Severity Rating (SSR). The SSR was the mean annual DSR for each voxel. We hypothesized that the SSR over the fire season would represent seasonal conditions, such as "a dry year".

We expected the five DSR indices to be correlated since they were derived from the same data however each index described the temporal pattern of fire weather in a slightly different way (Figure 4-2). Our goal was to select the index, or set of indices that parsimoniously described variation in lightning fire initiation.

Statistical models

Previous work with a subset of these data (Krawchuk and Cumming in preparation) suggested the selection of the zero-inflated negative binomial (ZINB) model (Simon Jackman's [pscl] package) for regression analysis. The distribution of raw data (Figure 4-1) and residuals from preliminary regression analyses supported this distribution over a zero-inflated Poisson, negative binomial, or Poisson model of errors. The mixture distribution for ZINB was selected over the conditional hurdle since all indices that contributed to the negative binomial component of the model should also be considered in describing the zero values. We could not include all variables in a true global model since correlation among DSR indices resulted in convergence failure. Instead, we screened the five candidate variables independently in both the binomial and negative binomial component of the model using the Wald statistic and a critical $\alpha = 0.05$ to provide a sub-global set of variables that explained the most variation in initiation. We selected a best model from this sub-global model using backward stepwise selection. We used the Akaike's Information Criterion (AIC) for selection among candidate models and tested for statistical significance from the null hypothesis ($\beta=0$) for each parameter estimate using the Wald statistic and a critical $\alpha = 0.05$

in the spirit of pluralism between frequentist and information theoretic methods (Stephens et al. 2005). Residual analysis was used to assess distribution assumptions of the model (Lambert 1992; Cameron and Trivedi 1998). We used the differences between the estimated and empirical probabilities of initiation to assess how well the model described variation of initiation counts based on Lambert (1992). Goodness of fit was assessed using a parametric bootstrap (Efron and Tibshirani 1998) on two metrics, the sums of squares (SS) and total regional count of initiations (RCI) over the study period using parameter estimates from the best model.

Results

The DSR-sequence count and SSR variables were selected in the best model of lightning fire initiation (Table 4-1), the former describing the binomial (excess zero) component, the latter describing the counts per voxel. The distribution of these two variables over the 19 years of the study data are illustrated in Figure 4-3 using box-and-whisker plots. Both variables show their highest distributions and values in 1998. Raw residuals supported the specification of the model. Parametric bootstrap indicated a good fit of the data to the model for both the sums of squares ($SS = 3961.6$, $p = 0.13$, $n = 1000$) and regional counts of initiation ($RCI = 2607$, $p = 0.51$, $n = 1000$). The difference between the estimated and empirical probabilities for initiation values was small showing no systematic shortcomings in our predictions of initiation.

Future patterns of fire weather and lightning fire initiation using the CRCM

Methods.

We predicted a distribution of initiation counts for each landscape by generating 1000 random deviates from a ZINB model with parameter estimates from the best model of observed initiation (Table 4-1) and simulated DSR-sequence count and SSR for the years 1975-1985 ($1 \times CO_2$ scenario), 2040-2049 ($2 \times CO_2$) and 2080-2089 ($3 \times CO_2$). The weather data used to calculate the two indices were acquired from the second policy run of the Canadian Regional

Climate Model (CRCM; Laprise et al. 1998) nested within the Canadian Global Climate Model (AGCM2). Data were prepared and calibrated by the Canadian Forest Service and these methods are described fully in Wotton (2001). The CRCM produces simulated weather data at six hour intervals for 45 km² grid boxes across western Canada. Existing methods were used to convert grid box averages of the CRCM to point value estimates used to calculate daily DSR values from solar noon weather data. We interpolated these point estimates of DSR across the study region using the inverse-distance methods described earlier. We compared the distribution of SSR and DSR-sequence count representing simulated and empirical 1 × CO₂ conditions (1975–1985 and 1983–2001, respectively) and also focused on the overlapping sequence of 1983–1985 from both data sets. We also compared the distribution of SSR and DSR-sequence count for the three scenarios of simulated data (1975–1985, 2040–2049, 2080–2089).

Results

Simulated SSR and DSR-sequence count indices of 1 × CO₂ conditions had a broader variance than observed indices (Figure 4-4), however the overall relationship suggested the 1 × CO₂ simulation was representative of recent weather conditions, as suggested by Wotton (2001). The median of the simulated indices for 2040–2049 (2 × CO₂) and 2080–2089 (3 × CO₂) scenarios showed the marked increase through time over both simulated and empirical indices for 1 × CO₂ however the upper whiskers of DSR-sequence count were highest in the 2 × CO₂ scenario (Figure 4-4). The generated frequency distribution of initiation increased through time in association with increasing fire weather conditions (Figure 4-5). The ratio of both the median and mean of initiations per voxel in the 2 × CO₂: 1 × CO₂ and 3 × CO₂: 1 × CO₂ scenarios were 1.5 and 1.8, respectively.

Area burned based on simulated changes in future climate

Methods

Spatio-temporal patterns of initiation are one component of the lightning fire regime of a region. The initiation declares a location from which a fire has the potential to grow, but not all initiations grow to be large fires due to fire suppression activity (Cumming 2005; Arienti et al. 2006) or because environmental factors such as changes in vegetation or moisture extinguish the fire. We evaluated the ecological effect of increased initiations expected in the future with existing empirically derived estimates of the number of initiations that escape to be large fires, and the size distribution of such fires. We used the term 'escaped' to describe a fire that reaches a size greater than nine ha. Cumming (2005) and Arienti et al. (2006) have estimated the probability of fire escape conditional on initiation for this study area. We selected the probability of escape, $p_{esc} = 0.084$, estimated from data between 1971 and 1993. Cumming (2001a) estimated the size distribution for fires greater than nine ha from events recorded in the study region between 1961 and 1998. Fire size is best described as a truncated exponential distribution with shape parameter, $\sigma = 2.56$ and the truncation point $b = 11.18$ corresponding to a maximum size of 643 000 ha (Cumming, 2001a).

We used the estimates of escape probability and the fire size distribution with our predictions of initiation to simulate annual area burned by lightning fire under $1 \times \text{CO}_2$, $2 \times \text{CO}_2$, and $3 \times \text{CO}_2$ conditions. In this study we assumed that escape probability and the fire size distribution were time- and space-invariant though data suggest that changes in fire operations implemented in 1983 may have reduced p_{esc} to 0.064 (Cumming 2005). The estimates selected for p_{esc} and the fire size distribution will influence the absolute results of area burned to be calculated below but not the relative value (i.e., trend through time). Fire suppression effectiveness has also likely influenced the fire size distribution over time, but the long-term fire size distribution provided by data from as far back as 1961 captures important temporal variability and acknowledges the potential for fire policy to adapt in the future to allow fires to burn more freely in some areas.

Cumming (2001a) also showed that vegetation influences fire size however we did not include this information in the conceptual framework of this study.

A regional annual sum of initiations was calculated for each of the 1000 permutations of the initiation counts generated for each voxel. We multiplied each regional annual sum of initiations by p_{esc} to produce the number of escaped fires (F_{esc}) then generated F_{esc} random samples from the truncated exponential fire size distribution. The cumulative area was used to calculate the regional annual area burned for each of the 1000 iterations. We used the median of each annual distribution in a linear regression analysis to test for a significant trend in area burned through time. These data were time slices rather than transient values however the hypothesis of a positive trend through time is still valid. The estimates of standard error were corrected using sandwich estimators to account for temporal autocorrelation in residuals. We calculated the mean of the annual medians in each scenario (e.g., mean of the annual medians for 1975 – 1985, 2040 – 2049, 2080 – 2089) to quantify the ratio in area burned resulting from expected changes in initiation frequency.

Results

The distributions of area burned suggest an increase through time with substantial variation within and among years (Figure 4-6). There was a significant, positive, linear trend in median area burned over time ($\beta = +1854.6$, s.e. = 271.0, $n = 31$, $p = 7.7 \times 10^{-12}$). The ratio of the mean median area burned was 1.9 for the $2 \times \text{CO}_2 : 1 \times \text{CO}_2$ scenario and 2.6 for the $3 \times \text{CO}_2 : 1 \times \text{CO}_2$ scenario.

Discussion

The predicted future rate of lightning fire initiation increased in central-eastern Alberta due to changes in fire weather conditions simulated by the CRCM. The increased frequency of fire weather conditions conducive to initiation provides a conservative estimate of future lightning fire activity given that calculations by Price and Rind (1994) based on climate simulations suggest

an increase in lightning activity for the United States and the northern hemisphere. In California, Fried et al. (2004) suggest that climate change will increase the intensity and spread rate of fires such that they are more likely to escape initial containment efforts and burn larger areas. Similarly Arienti et al. (2006) showed that the probability of fire escape increased under more extreme observed fire weather conditions (measured as a spread index). Our simulations suggested that the 1.5-fold increase in initiation between the $1 \times \text{CO}_2$ and $2 \times \text{CO}_2$ scenarios, and 1.8-fold increase between $1 \times \text{CO}_2$ and $3 \times \text{CO}_2$ scenarios correspond to 1.9-fold and 2.6 fold increases in annual area burned, respectively, without accounting for changes in the probability of fire escape. The magnitude of these prediction correspond with those calculated by Flannigan et al. (2005) who estimated the relationship between historical measures of area burned (lightning and human-caused fires) and weather data then predicted future area burned using these parameter estimates and weather data provided by the Hadley and Canadian GCMs. The Hadley and Canadian GCM scenarios predict 3.4 and 1.1-fold (respectively) increases in area burned for this study region in a $3 \times \text{CO}_2$: $1 \times \text{CO}_2$ scenario (Flannigan et al. 2005). Tymstra et al. (in review) showed a 1.2 – fold increase in area burned between $2 \times \text{CO}_2$ and $1 \times \text{CO}_2$ conditions and a 1.39 -fold increase between $3 \times \text{CO}_2$ and $1 \times \text{CO}_2$ conditions in a simulation study of fire behaviour under two climate scenarios in a study region adjacent to ours. The magnitude of these estimates (Flannigan et al. 2005; Tymstra et al. in review) of future area burned are equivalent to those we generated purely from the initiation phase of fire activity, independent of changes in the length of fire season, cause, escape probability, or fire size. This suggests that an integrated, hierarchical estimate of future fire activity might be very different from those predicted for any one process. Wotton and Flannigan (1993) predicted a 16% increase in the length of fire season for this region in a $2 \times \text{CO}_2$ versus $1 \times \text{CO}_2$ scenario. Any increase in length of fire season to our study would likely result in a further increase to suitable fire weather and an even greater increase in area burned.

Predictions made by simulation models about future weather and fire events across Canada vary as a function of simulator and parameter selection. We

demonstrated that an increase in initiations rate of less than 2-fold within the currently defined fire season can result in a substantial increase to area burned. Our results counter suggestions that changes in initiation rate may not be ecologically meaningful since a small number of initiations are typically responsible for the majority of area burned (Weber and Stocks 1998). Environmental conditions influence the likelihood of a fire initiation growing to be large (Arienti et al. 2006), but an increased number of initiations results in more chances to experience suitable conditions for large fires to occur.

Lightning fire initiation is strongly regulated by vegetation type as well as weather conditions (Krawchuk et al. 2006). The annual probability of initiation in a landscape varies with disturbance history and the relative composition of coniferous and deciduous species. In the boreal mixedwood, initiation probability is low in recently burned and aspen-dominated areas and higher in black or white spruce-dominated (*Picea mariana* and *P. glauca*) and recently harvested areas, likely due to variation in fuel composition (Krawchuk et al. 2006; Krawchuk and Cumming in preparation). Arienti et al. (in preparation) have found evidence for increased lightning fire initiation along linear features such as roads, seismic lines or pipelines that are vegetated by native and non-native grasses. Fire behaviour (such as spread) is similarly influenced by vegetation and weather (Forestry Canada Fire Danger Group 1992; Cumming 2001b; Hély et al. 2001; Bergeron et al. 2004) such that future patterns of initiation and behaviour will be regulated not only by changes in weather, but also by changes in vegetation. Current management decisions will direct future vegetation composition via forest harvesting and wildfire suppression. Expected changes in fire activity due to weather conditions, such as those presented herein, need to be integrated with vegetation models to further understand the complex outcome of these interacting factors.

Table 4-1. Parameter estimates for the best ZINB model quantifying lightning fire initiation ($\theta = 0.67$). The ΔAIC was calculated as the change in value when the variable was removed from this model.

Variable	Estimate	s.e.	p-value	ΔAIC
β_0 (binomial)	+1.06	0.16	5.60×10^{-11}	113
DSR-sequence count	-0.34	0.047	1.31×10^{-13}	
β_0 (negative binomial)	-1.53	0.11	1.63×10^{-45}	10
SSR	+0.13	0.038	2.14×10^{-3}	

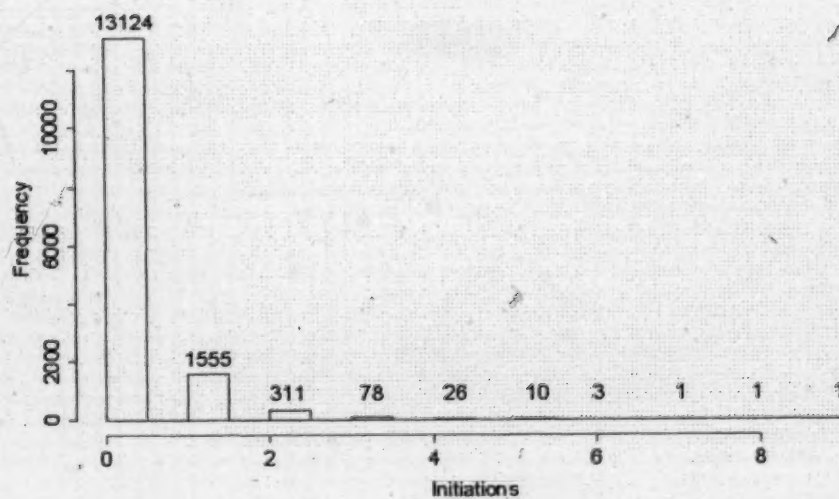


Figure 4-1. The frequency distribution of observed initiations per voxel between 1983 and 2001.

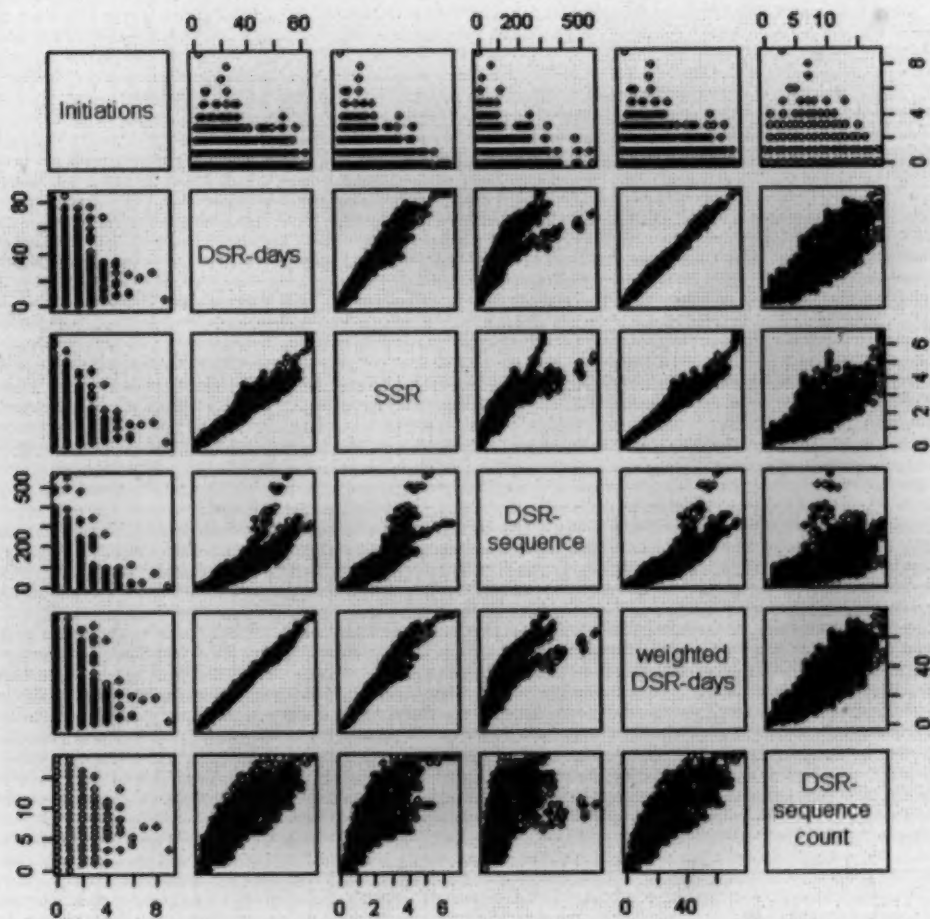


Figure 4-2. The pair-wise correlation of fire initiation and the indices of fire weather used to describe variation in initiation. The Daily Severity Rating (DSR) was used to create five different indices describing different hypotheses about the relationship between initiation and weather conditions.

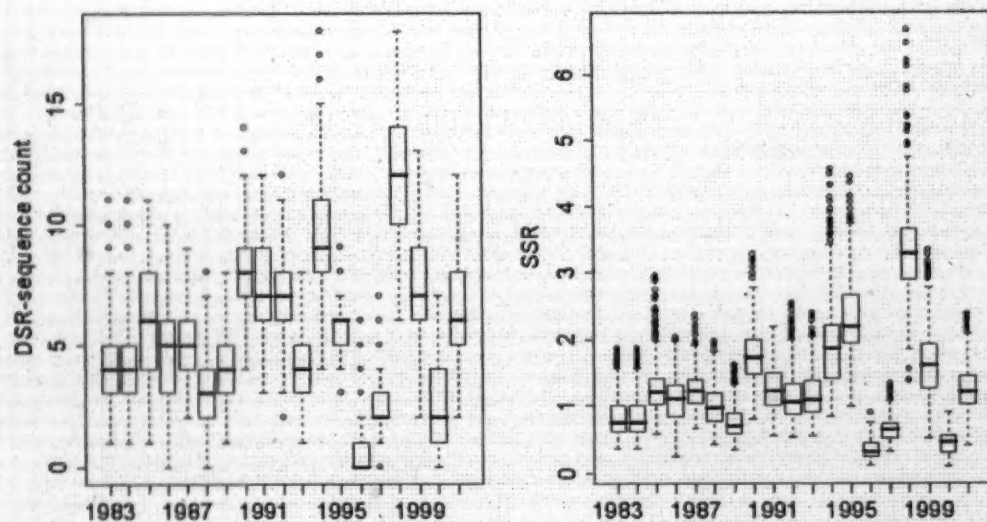


Figure 4-3. The annual distribution of empirical DSR-sequence counts (Daily Severity Rating) and SSR (Seasonal Severity Rating) values per voxel between 1983 and 2001 illustrated by box plots.

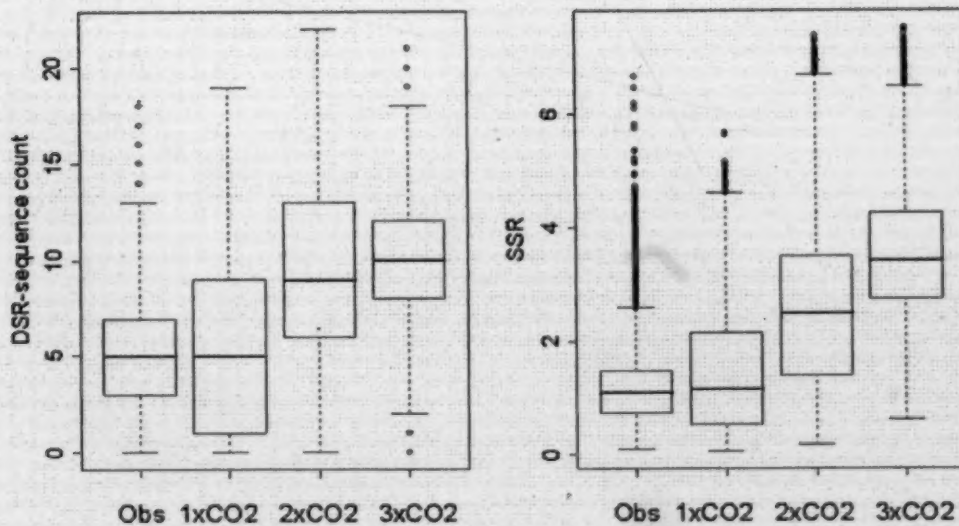


Figure 4-4. The boxplot distribution of DSR-sequence count (Daily Severity Rating) per voxel for 1983-2001 (Obs $1 \times \text{CO}_2$), 1975-1985 (simulated $1 \times \text{CO}_2$), 2040-2049 (simulated $2 \times \text{CO}_2$) and 2080-2089 (simulated $3 \times \text{CO}_2$). The distributions in SSR (Seasonal Severity Rating) per voxel are presented for the same time sequences.

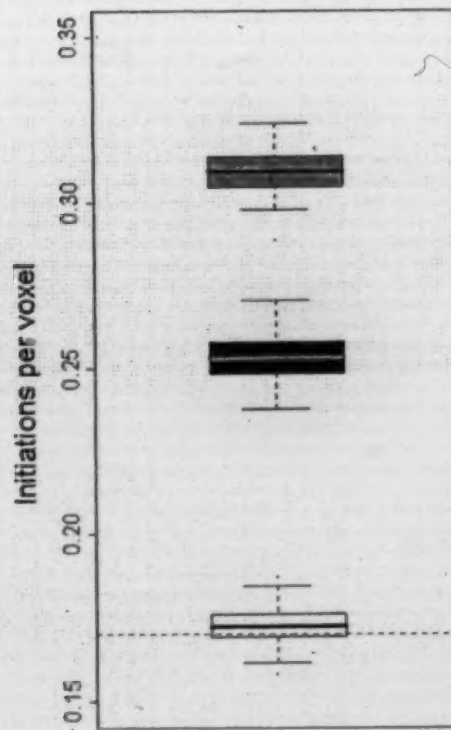


Figure 4-5. The box plot distribution of mean predicted initiations per voxel for 1975 -1985 (white inter-quartiles; 1 × CO₂), 2040-2049 (black inter-quartiles; 2 × CO₂) and 2080-2089 (grey inter-quartiles; 3 × CO₂) generated from 1000 random deviates from the zero-inflated negative binomial model. The dashed horizontal line illustrates the observed mean number of initiations per voxel for 1983 – 2001.

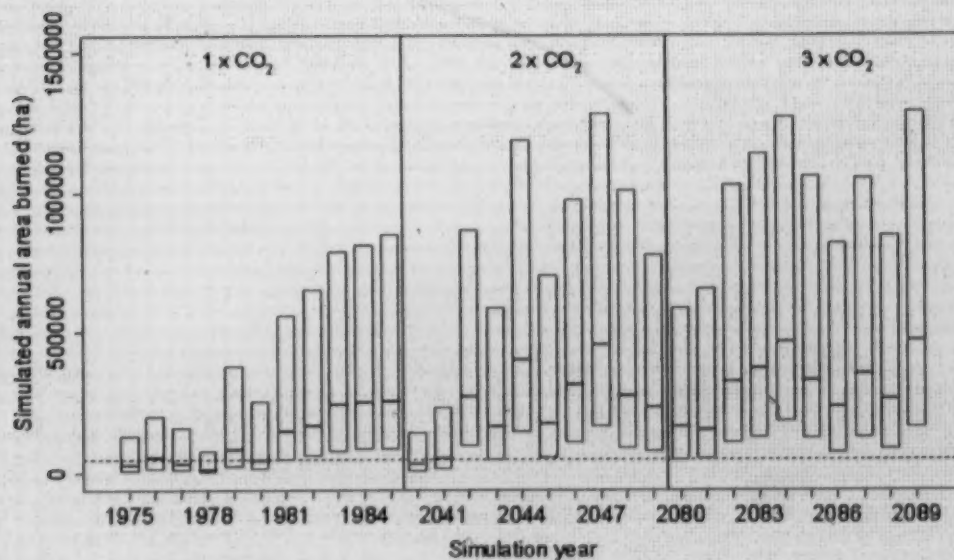


Figure 4-6. The box plot distribution of 1000 iterations of annual area burned illustrated omitting whiskers and outliers. Solid vertical lines show the split for years representing $1 \times \text{CO}_2$ (1975-1985), $2 \times \text{CO}_2$ (2040-2049) and $3 \times \text{CO}_2$ (2080-2089). The dashed horizontal line shows the area representing a 0.41% annual burn rate calculated from 54 years of regional fire history data by Cumming (1997).

Literature Citations

- ASRD. 2005. Historical wildfire database.
http://www.srd.gov.ab.ca/wildfires/fpd/wi_hdhwd.cfm; accessed 10 September 2005.
- Arienti, M.C., S.G. Cumming and S. Boutin. 2006. Empirical models of forest fire initial attack success probabilities: the effects of fuels, anthropogenic linear features, fire weather, and management. *Canadian Journal of Forest Research* **36**:3155-3166.
- Anderson, K., and P. Englefield. 2001. Quantile characteristics of forest fires in Saskatchewan. Pages 9-16 in *Proceedings of the Fourth Symposium on Fire and Forest Meteorology*, 13-15 November, 2001, Reno Nevada. American Meteorological Society, Boston, Massachusetts.
- Bergeron, Y., S. Gauthier, M.D. Flannigan and V. Kafka. 2004. Fire regimes at the transition between mixedwood and coniferous boreal forest in northwestern Quebec. *Ecology* **85**:1916-1932.
- Cameron, A.C., and P.K. Trivedi. 1998. *Regression Analysis of Count Data*. Cambridge University Press, Cambridge.
- Clark, J.S. 1998. Why trees migrate so fast: Confronting theory with dispersal biology and the paleorecord. *American Naturalist* **152**:204-224.
- Cumming, S.G. 1997. *Landscape dynamics of the boreal mixedwood forest*. Dissertation, University of British Columbia. Vancouver, B.C. Canada
- Cumming, S.G. 2001a. A parametric model of the fire-size distribution. *Canadian Journal of Forest Research* **31**:1297-1303.
- Cumming, S.G. 2001b. Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn? *Ecological Applications* **11**:97-110.
- Cumming, S.G. 2005. Effective fire suppression in boreal forests. *Canadian Journal of Forest Research* **35**:772-786.
- Efron, B. and R.J. Tibshirani. 1998. *An Introduction to the Bootstrap*. Chapman and Hall/CRC, New York.
- ESRI. 2005. ArcGIS 9. Redlands, CA.
- Flannigan, M.D., Y. Bergeron, O. Engelmark and B.M. Wotton. 1998. Future wildfire in circumboreal forests in relation to global warming. *Journal of Vegetation Science* **9**:469-476.
- Flannigan, M.D. and J.B. Harrington. 1988. A study of the relation of meteorological variables to monthly provincial area burned by wildfire in

- Canada (1953-80). *Journal of Applied Meteorology* 27:441-452.
- Flannigan, M.D. and C.E. Van Wagner. 1991. Climate change and wildfire in Canada. *Canadian Journal of Forest Research* 21:66-72.
- Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner and B.J. Stocks. 2005. Future area burned in Canada. *Climatic Change* 72:1-16.
- Fried, J.S., M.S. Torn and E. Mills. 2004. The impact of climate change on wildfire severity: a regional forecast for northern California. *Climatic Change* 64: 169-191.
- Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian Forest Fire Behavior Prediction System. Forestry Canada Information Report ST-X-3, Ottawa.
- Greene, D.F. and E.A. Johnson. 1999. Modelling recruitment of *Populus tremuloides*, *Pinus banksiana*, and *Picea mariana* following fire in the mixedwood boreal forest. *Canadian Journal of Forest Research* 29:462-473.
- Hély, C., M.D. Flannigan, Y. Bergeron, Y. and D. McRae. 2001. Role of vegetation and weather on fire behaviour in the Canadian mixedwood boreal forest using two fire behaviour prediction systems. *Canadian Journal of Forestry Research* 31:430-441.
- Historical Wildfire Database. 2005. Alberta Sustainable Resource Development, Alberta Government. http://www.srd.gov.ab.ca/wildfires/fpd/wi_hdhwd.cfm. Cited 3 Aug. 2005.
- Hogg, E.H. and R.W. Wein. 2005. Impacts of drought on forest growth and regeneration following fire in southwestern Yukon, Canada. *Canadian Journal of Forest Research* 35:2141-2150.
- IPCC. 2001. *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, and Johnson CA (eds) Cambridge University Press, Cambridge, United Kingdom
- Jasinski, J.P.P. and S. Payette. 2005. The creation of alternative stable states in the southern boreal forest, Quebec, Canada. *Ecological Monographs* 75:561-583.
- Johnson, E.A., K. Miyanishi and S.R.J. Bridge. 2001. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. *Conservation Biology* 15:1554-1557.
- Johnstone, J.F. and F.S. Chapin. 2006. Fire interval effects on successional trajectory in boreal forests of northwest Canada. *Ecosystems* 9:268-277.

- Johnstone, J.F. and F.S. Chapin. 2003. Non-equilibrium succession dynamics indicate continued northern migration of lodgepole line. *Global Change Biology* 9:1401-1409.
- Krawchuk, M.A., S.G. Cumming, M.D. Flannigan and R.W. Wein. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. *Ecology* 87:458-468.
- Lambert, D. 1992. Zero-inflated Poisson regression, with an application to defects in manufacturing. *Technometrics* 34:1-14.
- Laprise, R., D. Caya, M. Giguere, G. Bergeron, G.J. Boer and N.A. McFarlane. 1998. Climate and climate change in western Canada as simulated by the Canadian Regional Climate Model. *Atmosphere-Ocean* 36:119-167.
- Li, C., M.D. Flannigan and I.G.W. Corns. 2000. Influence of potential climate change on forest landscape dynamics of west-central Alberta. *Canadian Journal of Forest Research* 30:1905-1912.
- McIntire, E.J.B., R. Duchesneau and J.P. Kimmins. 2005. Seed and bud legacies interact with varying fire regimes to drive long-term dynamics of boreal forest communities. *Canadian Journal of Forest Research* 35:2765-2773.
- Overpeck, J.T., D. Rind and R. Goldberg. 1990. Climate-induced changes in forest disturbance and vegetation. *Nature* 343:51-53.
- Paine, R.T. and S.A. Levin. 1981. Inter-tidal landscapes - disturbance and the dynamics of pattern. *Ecological Monographs* 51:145-178.
- Paine, R.T., M.J. Tegner and E.A. Johnson. 1998. Compounded perturbations yield ecological surprises. *Ecosystems* 1:535-545.
- Parisien, M.A., K. Hirsch, V. Kafka, B.J. Todd, N. Flynn and M.D. Flannigan. 2005. Fire behavior potential in central Saskatchewan under predicted climate change. *Prairie Adaptation Research Collaborative Document No. 05-01*.
- Pickett, S.T.A. and P.S. White. 1985. The ecology of natural disturbance and patch dynamics. Academic Press, Inc., San Diego, California.
- Price, C. and D. Rind. 1994. The impact of a 2x-CO₂ climate on lightning-caused fires. *Journal of Climate* 7:1484-1494.
- R Development Core Team. 2005. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
- Rowe, J.S. 1972. Forest Regions of Canada. Publication Number 1300. Canadian Forest Service, Ottawa, Ontario, Canada.

- Schoennagel, T., T.T. Veblen and W.H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. *Bioscience* **54**:661-676.
- Stephens, P.A., S.W. Buskirk, G.D. Hayward and C.M. Del Rio. 2005. Information theory and hypothesis testing: A call for pluralism. *Journal of Applied Ecology* **42**:4-12.
- Stocks, B.J., J.A. Mason, J.B. Todd, E.M. Bosch, B.M. Wotton, B.D. Amiro, M.D. Flannigan, K.G. Hirsch, K.A. Logan, D.L. Martell and W.R. Skinner. 2002. Large forest fires in Canada, 1959-1997. *Journal of Geophysical Research-Atmospheres* **107**:1-12.
- Strong, W.L. and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands, and Wildlife, Land Information Division, Resource Information Branch, Edmonton, Alberta
- Tymstra, C., M.D. Flannigan, O.B. Armitage and K. Logan. Impact of climate change on area burned in Alberta's boreal forest. *International Journal of Wildland Fire*. In review.
- Van Wagner, C.E. 1987. Development and structure of the Canadian Forest Fire Weather Index System. Forestry Technical Report 35. Canadian Forestry Service, Ottawa, Canada
- Weber, M.G. and M.D. Flannigan. 1997. Canadian boreal forest ecosystem structure and function in a changing climate: Impact on fire regimes. *Environmental Reviews* **5**:145-166.
- Weber, M.G. and B.J. Stocks. 1998. Forest fires and sustainability in the boreal forests of Canada. *Ambio* **27**:545-550.
- Wotton, B.M. 2001. Current and future forest fire occurrence and severity in Canada: Creation and validation of scenarios developed using the Regional Climate Model. Climate Change Action Fund Report S99-15-04.
- Wotton, B.M. and M.D. Flannigan. 1993. Length of the fire season in a changing climate. *Forestry Chronicle* **69**:187-192.

Chapter 5. The interaction between fire, vegetation, climate and management: Is there a biotic buffer to fire activity under climate change in the mixedwood boreal forest?

Introduction

Multiple scales of interacting biophysical processes contribute to boreal forest structure and changes in dominant drivers are likely to result in substantial alteration to the ecosystem. Climate exerts a strong, physical control on boreal systems (Bonan and Shugart 1989; Chapin et al. 2006) for example, variability in temperature and precipitation influence plant growth, moisture balance and disturbance activity (Zhang et al. 2000; Gillett et al. 2004; Hogg and Bernier 2005; Kasischke and Turetsky 2006; Chapin et al. 2006). Changes in climate are expected to alter weather patterns in northern and western regions of Canada more rapidly in the 21st century than has been observed in recent history (IPCC 2007). This climate warming is expected to have a profound and rapid influence on fire activity in the boreal forest because of the direct relationship between weather conditions and fire activity (Weber and Flannigan 1997).

A fire regime is a 'slow variable' (Chapin et al. 2006), generated by the interaction of vegetation, climate and ignition events, moderated by human activities including forest management and fire suppression (Quintilio et al. 1991; Weir et al. 2000; Cumming 2001a; Flannigan et al. 2005; Cumming 2005; Krawchuk et al. 2006). The mixedwood boreal forest region in western Canada is strongly shaped by wildfire (Weber and Stocks 1998). Predictions of future fire activity (e.g., area burned) have dominantly been generated from climate data alone (e.g., Flannigan et al. 2005) based on the argument that direct relationships between climate and fire will overshadow and outpace slower, or less important changes generated by alteration in vegetation structure (Weber and Flannigan 1997). An increase in fire activity has been projected from relationships between weather data and fire initiation (Krawchuk et al. in review (Chapter 4)), fire intensity (Parisien et al. 2005), area burned (Flannigan et al. 2005) and over an extended fire season (Wotton and Flannigan 1993) within the 21st century.

However the magnitude of near-future changes in the fire regime calculated in the absence of vegetation could be misleading (Gardner et al. 1996; Chapin et al. 2000; Hinzman et al. 2003; Schumacher and Bugmann 2006). Future changes in vegetation might result from changes in the effectiveness of fire suppression, or by landscape conversion resulting from human development and management. Characteristics of the vegetation community might change within the century in response to climate via dispersal events (Clark 1998) or changes in phenology (Beaubien and Freeland 2000), and within decades via changes in post-fire regeneration (Hogg and Bernier 2005; Hogg and Wein 2005) or shifts in the outcome of community competition (Suttle et al. 2007). Even without direct effects of climate on vegetation an increase in fire activity for several decades may decrease subsequent regional fire hazard (Gardner et al. 1996; Chapin et al. 2000) via an increased area of relatively non-flammable, early successional vegetation, reduced fine fuels (Quintilio et al. 1991; Schimmel and Granstrom 1997; Lavoie 2004), and increased fuel heterogeneity in the landscape. The direct effect of increasing fire weather conditions on future fire activity could be diminished as a function of this non-linear response. It is unclear whether the predicted rate of climate change is large enough to effectively trigger this type of negative feedback.

Weather and fuel condition contribute to fire activity according to their heterogeneity at different scales (Bonan and Shugart 1989; Schoennagel et al. 2004; Krawchuk et al. 2006). For example, daily weather conditions influence fire risk (Van Wagner 1987; Bessie and Johnson 1995; Wotton et al. 2003) and determine monthly (Flannigan and Harrington 1988) and inter-annual differences in fire activity (Krawchuk et al. 2006), whereas variation in climate over decades and centuries determines the range of biotic species susceptible to burning. The western mixedwood boreal forest is characterized by its mesic, mixedwood stands typically composed of white spruce (*Picea glauca* (Moench) Voss) and trembling aspen (*Populus tremuloides* Michx.). These stands develop through stages that often include succession from trembling aspen to white spruce, with initial stand condition, landscape context and disturbance history affecting this trajectory

(Chen and Popadiouk 2002; Albani et al. 2005; Johnstone and Chapin 2006a). The natural heterogeneity in these landscapes contributes to spatio-temporal patterns in fuel and fire hazard among years and decades (Quintilio et al. 1991; Cumming 2001a; Krawchuk et al. 2006). Aspen and spruce forest pose a strong contrast in fire hazard (Van Wagner 1977; Hély et al. 2000; Cumming 2001a; Krawchuk et al. 2006; and see Johnson et al. 1998). Lightning fire initiation is less likely to occur in moist, broad-leaved aspen forests than white spruce forest with abundant needle-leaves as ladder fuels, even under extreme weather conditions (Krawchuk et al. 2006). Wildfire behaviour is moderated in aspen stands (Van Wagner 1977; Quintilio et al. 1991; Hély et al. 2001; Hinzman et al. 2003) except under extreme weather conditions or during the short window of time in spring prior to leaf-out (Quintilio et al. 1991). The succession-driven change in fuel load implies an age-related fire hazard in mesic stands. The dynamics of a mesic mixedwood boreal forest stands are compatible with the adaptive cycle of ecosystem function (Gunderson and Holling 2002) with reorganization after fire, exploitation of available resources by pioneer deciduous species, slow conservation of nutrients and biomass in white spruce, then release as creative destruction generated by natural disturbance.

Anthropogenic activity is accelerating in Canada's boreal forests such that harvesting and other industrial disturbances will dramatically modify vegetation structure within the next century (Schneider et al. 2003). The majority of the mixedwood boreal region is under commercial forest tenure as are one half of Canada's forests (Smith and Lee 2000) and the interaction between fire and forest management is not well understood, especially in the context of climate change. Forest management in Alberta, Canada has a broad interest in emulating natural disturbance however wildfire and anthropogenic disturbances result in outcomes that differ from one another substantially (Song 2002; Krawchuk and Cumming *submitted* (Chapter 3)). Mixedwood boreal management generates landscapes composed of recently harvested and regenerating stands in addition to those originating from natural post-fire stand dynamics. Field experiments and simulation studies show that forest management can influence fire activity by

altering the amount and configuration of fuels (Quintilio et al. 1991; Stephens 1998; Hirsch et al. 2001; Song 2002; Gustafson et al. 2004; Krawchuk and Cumming *submitted* (Chapter 3)). In previous work (Krawchuk and Cumming *submitted* (Chapter 3)) we showed an increased probability of lightning fire initiation for 10 to 30 years in landscapes with increasing post-harvest area as compared to post-fire areas. This effect is likely due to abundant fine fuels from harvest operations intermixed with flammable dry and matted grass (Hirsch et al. 2001; Baxter 2004) despite the quick growth of deciduous vegetation that dominates stands after disturbance. Hirsch et al. (2001) studied the effect of forest management stand configurations on area burned as a potential method to reduce fire hazard within the current management policy in Alberta. The minimization of wildfire hazard on forest tenures via an alternative regeneration policy has not yet been considered. In fact, the current forest regeneration policy may contribute to fire activity in the region in a drive towards static states and sustainable wood volume. The current policy 'unmixes' the mixedwood forest by legislating forest companies to re-stock harvested white spruce stands immediately to white spruce *in lieu* of a successional transition from aspen to spruce over time - a natural pattern that may also minimize the likelihood of burning early on. There may be an opportunity to adapt to the fire risk presented by climate change through alternative regeneration policies that reduce the area of fire-prone, pure white spruce forest planted in the region.

The fire risk that climate change presents has spurred the forest industry to think closely about the effects of fire on merchantable forest landscapes (Li et al. 2000; Johnston et al. 2001). Integrated assessments exploring the future state of boreal forests are challenged by the complexity of incorporating positive and feedbacks between climate, vegetation and disturbance occurring at multiple spatial and temporal scales (Kittel et al. 2000; Chapin et al. 2000; Schumacher and Bugmann 2006). Existing studies of climate change, wildfire and forest management in the mixedwood boreal forest region are disconnected. We developed six scenarios for a simulation experiment to study the interaction of these three variables through the extension of an existing spatially explicit model

of mixedwood boreal forest dynamics, Tardis (Cumming and Armstrong 2001; Cumming and Armstrong 2004; Krawchuk and Cumming *submitted* (Chapter 3)). We developed extensions of the existing Tardis model to provide: i) an empirically derived wildfire module sensitive to climate and forest composition, and ii) an alternative regeneration harvest scheduling module as. Climate projections were quantified using data from the Canadian Regional Climate Model (CRCM; Laprise et al. 1998). The experiment included a 250 year simulation interval within a 9 million ha study region in central eastern Alberta. We tested three hypotheses: First, wildfire activity would increase with climate change, but negative feedback between the increased area of forest in post-fire, early successional stands, and subsequent wildfire activity would reduce the change in regional area burned predicted from models that omit the regulatory effects of vegetation on fire activity (e.g., Flannigan et al. 2005; Krawchuk et al. in review (Chapter 4)). Second, fire activity would increase locally in harvested areas for a short period as a result of increased fire initiation in the decades immediately after harvest operations (Krawchuk and Cumming *submitted* (Chapter 3)), but that net regional fire activity (initiation and area burned) would be reduced in forest harvest scenarios due to the area of young and mature aspen-dominated mesic forest generated by harvest operations. Third, the implementation of an alternative forest regeneration policy that increased the area of young aspen dominated mesic forest and decreased the area of young white spruce-dominated mesic forest would reduce long-term, regional area burned.

Methods

Study area, ecology and management

The simulation area was 93 500 km² of mixedwood boreal forest (Rowe 1972) in central-eastern Alberta, Canada (Figure 5-1). This mixedwood forest is the most productive and diverse boreal ecosystem in North America (Chen and Popadiouk 2002). The historical regional climate is continental with mean winter and summer temperatures of -13.2°C and +13.5°C, respectively and total precipitation in summer and winter in the range of 240 mm and 64 mm, respectively (Strong and Leggat 1992). Glacial movement through the region has resulted in rolling morainic deposits on the uplands and smoother glacio-lacustrine deposits on the lowlands (Rowe 1972). Elevation undulates from approximately 200 to 1000 m. Forest tree communities are stratified by soil moisture regime (Strong and Leggat 1992). Upland mesic sites are dominated by mixtures of trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.) or white spruce (*Picea glauca* (Moench) Voss). Community development at these mesic sites depends on variation in disturbance history, composition, and spatial context (e.g., stochastic availability of propagules from neighbouring stands). Paper birch (*Betula papyrifera* Marsh) is present but pure stands are rare. White spruce and balsam fir (*Abies balsamea* (L.) Mill) dominate older stands but pure stands are not currently abundant. Xeric sites are vegetated by jack pine (*Pinus banksiana* Lamb), sometimes in association with aspen. Moist, poorly drained sites are vegetated by black spruce (*Picea mariana* (Mill.) B.S.P.). There are extensive wetlands including open areas of bog, fens, swamps and marshes. Tamarack (*Larix laricina* (Du Roi) Koch) is the dominant tree species found in wetlands. The region's vegetation composition as classified for this study is presented in Table 5-1.

The area burned by lightning- and human-caused fires has contributed to the heterogeneous species and age structure of the region. Fire characteristics vary by intensity, seasonality, and type such that the severity of fire is heterogeneous within and among events. Lightning fires burn the majority of the area but the frequency of human-caused fires is increasing (Alberta Sustainable Resource

Development 2006), likely due to increased access. Arienti et al. (in review) show that the frequency of lightning fire initiation increases with the density of roads, probably due to the flammable grasses that grow along road verges. Annual area burned by lightning fires in the study area since 1961 ranges between ~ 4 and 700 000 ha (Alberta Sustainable Resource Development 2006). A small number of large fires are responsible for the majority of area burned (Weber and Stocks 1998). Cumming (1997) estimated the mean annual rate of burn by lightning and human caused fires (corrected for suppression effects) between 1940 and 1993 at 0.41 %. Fire management has occurred in the study region since 1968 (Murphy 1985). Fire suppression is successful in containing the growth of small fires, however large fires are frequently un-approachable during their large runs during extreme fire conditions.

The forest industry harvests deciduous and coniferous trees from the region, but focuses on the removal of aspen (deciduous) and white spruce (coniferous) from mesic stands. Aspen is the dominant species harvested by the primary commercial operator, Alberta-Pacific Forest Industries Inc. (ALPAC); operations began in 1994. Small scale operators have harvested conifer since the 1950s, including white spruce, jack pine and black spruce. The following details of forest operations were provided by ALPAC (Alberta-Pacific Forest Industries Inc. 2004; SmartWood 2005). The majority of timber is removed using a two-pass system of clear-cutting using feller-bunchers and grapple skidders, clearing approximately 14 000 ha annually. In small areas ALPAC has experimented with shelterwood harvesting of mesic sites where the removal of deciduous overstory can release conifer from the understory. Timber is processed at the road-side to remove branches and tops, leaving the majority of slash along these verges. Different techniques have been used to configure harvest slash since industrial operations began, but harvest slash is not systematically burned (Baxter 2004). Silviculture is based on a suite of provincial regeneration rules. Deciduous sites rely on natural regeneration and coniferous sites are typically planted to white spruce or pine, according to policy guidelines.

Oil and gas development in the region results in additional forest removal (Schneider et al. 2003). The cumulative footprint of seismic lines and well sites is comparable to forest harvesting. However the configuration of this footprint is markedly different from harvesting; e.g., linear features have a width of only two to seven metres. The influence of these linear features on fire activity is uncertain however Arienti et al. (2006) show they do not systematically influence the initial suppression or growth of fires.

Simulation experiment

We compared the outcome of six scenarios within a two factor experimental design to quantify the interaction between forest composition, wildfire, forest management and climate change. The disturbance factor was simulated with three levels: fire only (natural disturbance), fire and forest management (current management policy), fire and alternative forest management (alternative regeneration policy). The current management scenarios followed current harvesting and regeneration policy for the region. The alternative management scenarios used modified post-harvest regeneration rules designed to reduce fire hazard by minimizing the amount of young white spruce dominated mesic forest subject to sustaining harvest volumes. The climate factor included two levels, a constant climate and a changing climate. Scenarios of constant climate conditions were simulated with $1 \times \text{CO}_2$ conditions from the CRCM (Laprise et al. 1998). Climate change was simulated with CRCM data at $1 \times \text{CO}_2$, $2 \times \text{CO}_2$ and $3 \times \text{CO}_2$ conditions.

The six scenarios represented the following conditions: [1] no climate change and no forest management (NCC-NFM); [2] no climate change and current forest management policy (NCC-FM); [3] no climate change and alternative forest management policy (NCC-AM); [4] climate change and no forest management (CC-NFM); [5] climate change and current forest management policy (CC-FM); [6] climate change and alternative forest management policy (CC-AM). Each scenario was simulated 100 times to capture a distribution in responses generated by stochastic elements of the Tardis model

and the climate data. All scenarios began with a 31 time step (155 year) burn-in period under constant $1 \times \text{CO}_2$ conditions and scenario-dependent disturbance (harvesting and/or fire) regime, followed by a 250 year simulation. The first 27 time steps (135 years) following the burn-in period were used as the experimental interval. The extent of the interval was selected to coincide with the time series of climate data available from the CRCM. The remaining 23 time steps (115 years) were used to explore the system under a continued, constant $3 \times \text{CO}_2$ climate.

Tardis simulation model

Land base

Tardis is a spatially explicit stochastic model of mixedwood boreal forest dynamics and management developed and calibrated for this region (Cumming and Armstrong, 2001; 2004; Cumming and Vernier 2002). The model operates at a spatial resolution of $\approx 10\,000$ ha landscapes and temporal resolution of five years. Tardis simulates the interactive processes of wildfire, harvesting, regeneration and succession. Landscape units were initialized with variables describing vegetation composition using aspatial stand tables from a regional forest inventory (Anonymous 1985). The original photography for the inventory was collected between 1970 and 1984 (Anonymous 1985). The inventory described the structure of forested and non-forested stands/areas. Stand attributes were updated to the year 2000 to reflect changes from fire, harvest and development. Forest stand attributes included species composition, density, age, and details of recent disturbances (date, disturbance type and effect). These fields were used to classify stands into ecologically meaningful cover types for the fire, harvest, regeneration and stand dynamics modules described below. Within landscapes, stand attributes change over time as a function of simulated disturbance and stand dynamics. Vegetation dynamics within the model maintain currently existing cover types, and disturbances (fire or harvesting) re-set the successional trajectory within forest cover types and via regeneration rules. New cover types were not generated in response to climate or disturbance interval, this was not a dynamic vegetation model (e.g., Rupp et al. 2002; Schumacher and

Bugmann 2006). Non-forested cover types such as peatland, water or anthropogenic development were invariant throughout the simulation.

Fire and climate

Wildfire was simulated by four independent stochastic processes: initiation, escape, final size, and effect (Cumming and Armstrong 2004). The number of fires initiated per time step and landscape was simulated based on an additive function of vegetation (landscape) composition and climate conditions using a zero-inflated negative binomial model (Table 5-2). The initiation model was developed for this application based on methods and empirical data presented in Chapter 3 (Krawchuk and Cumming (submitted)) and Chapter 4 (Krawchuk et al. (in review)) and was fit using historical vegetation, weather and fire data over the period 1994 to 2001. Vegetation covariates were calculated using a classification of the inventory data (Table 5-1) adapted for previous models (Krawchuk et al. 2006 (Chapter 2); Krawchuk and Cumming submitted (Chapter 3)). To model climate effects we required an annual landscape-scale index of fire weather conditions (VanWagner 1987) conducive to lightning fire initiation that could be derived from historical data and applied with climate data from the CRCM. We used the DSR-sequence count variable (Krawchuk et al. in review (Chapter 4)) which describes spatio-temporal patterns in fire weather conditions by the number of times in a year when the Daily Severity Rating (DSR; Van Wagner 1987) exceeds three for a sequence of two or more consecutive days during the fire season of May 1 to September 30 (Figure 5-2). Observed, historical values were calculated for each landscape and year from interpolated daily fire weather indices (Krawchuk et al. in review (Chapter 4)). We selected a parsimonious model for initiation where initiation increased with DSR-sequence count, the area of white spruce dominated mesic forest, black spruce dominated forest and forest harvested within the last 10 years. Initiation decreased with the area of mature (30 – 90 years) aspen-dominated mesic forest and forests burned within the last 30 years (Table 5-2).

Simulations used DSR-sequence count data prepared from the CRCM (Laprise et al. 1998) from interpolated gridded fire weather indices (Figure 5-2) validated by Wotton (2001), using methods presented in Krawchuk et al. (2006). We calculated DSR-sequence count for each landscape in each year of climate data. The CRCM contained data for three time slices representing $1 \times \text{CO}_2$ (1975 - 1985), $2 \times \text{CO}_2$ (2040 - 2049) and $3 \times \text{CO}_2$ (2080 - 2089). Scenarios with no climate change (i.e., current climate) used the $1 \times \text{CO}_2$ data. Climate change scenarios used restricted random sampling of annual data grids from all time slices to provide a transient stream of climate indices. The climate data were calibrated to the time steps of the simulation using rules in Table 5-3. For example, time steps between one and five (inclusive) used annual DSR-sequence count data from five random draws (with replacement) from the available grids representing $1 \times \text{CO}_2$ conditions. For time steps six and seven, four draws were made from the $1 \times \text{CO}_2$ conditions and one draw from $2 \times \text{CO}_2$ conditions, etc. (Table 5-3). At each time step vegetation covariates for each landscape were calculated from Tardis stand attributes, and DSR-sequence counts were obtained from the time series of climate variables. We took the sum of five independent random deviates sampled from the zero-inflated negative binomial model to calculate initiation counts for each five year time step.

Fire initiations that exceed a threshold size are said to have escaped (Cumming 2005). For each time step and landscape the total number of escapes was sampled from a binomial distribution (p_e, N_i) where p_e is the probability of escape and N_i is the total initiation count. We assumed $p_e = 0.12$ to represent historical rates and include the current effectiveness of fire management (Cumming 2005). We ignored local effects of vegetation and fire weather on fire escape (Arienti et al. 2006) though changes in climate, vegetation and development may alter escape probability in the future. Cumming (2005) showed little change in the probability of escape as a function of increasing numbers of initiations in the region under current fire suppression strategies.

Final sizes of escaped fires were sampled from a truncated exponential distribution developed for the study region from regional, empirical data from

1980 to 1998 (Cumming 2001b). The largest fire recorded during this interval was $\approx 428\,000$ ha (Cumming 2001b). Though it is a short time interval, no fires larger than those recorded during this time have occurred in the region since 1931 (Delisle and Hall 1987). The truncation point allows a maximum fire size of 643 000 ha. The distribution's shape parameter is a function of landscape vegetation composition (Cumming 2001b). In the original formulation, expected fire size increased with the proportional area of pine forest and decreased with the area of recently disturbed forest. Burned and harvested areas were not distinguished, but burned areas predominated. Fire behaviour differs among burned and harvested fuels (Alexander 1982; Quintilio et al. 1991; Forestry Canada Fire Danger Group 1992; Stephens 1998; Kurt Frederick, Alberta Sustainable Resource Development, pers. comm.) and we have since shown that the two disturbances have opposite effects on fire initiation (Krawchuk and Cumming submitted (Chapter 3)). Therefore we only included burned forest in the disturbed category, and considered harvested forest to be neutral at this time. Similarly, there is convincing evidence that aspen forest moderates fire intensity and spread in most weather conditions (VanWagner 1977; Quintilio et al. 1991; Hély et al. 2001). Thus increasing amounts of aspen forest would be expected to reduce mean fire size locally. Accordingly, we included mature aspen dominated forests in the disturbed class when sampling fire size.

Fires do not burn all vegetation types equally, the composition of area burned by individual fires is strongly related to the composition of the landscape in which the fire is initiated (Cumming 2001a). We modelled this aspect of fire effect using a multivariate regression model adapted from Cumming (2001a). We do not model variation in fire severity, assuming that all fires are stand-replacing. This is not true, but stand replacing crown fires and intense surface fires burn the majority of area in the boreal forest (Bonan and Shugart 1989). Though human caused fires are increasingly frequent lightning fires are responsible for the majority of area burned by wildfire; we did not include human caused fires in our simulations. Fire regime indicators such as initiation or area burned are an

emergent pattern generated through changes in land base composition, forest management, stand dynamics and spatially variable patterns of fire weather.

Harvest

Forest harvest was simulated with a single harvest agent responsible for forest management and timber supply to 18 mills in the region. This arrangement emulated existing forestry facilities and practices (see Study area, ecology and management) with a global forest management policy presented by Cumming and Armstrong (2001; 2004). This policy centralizes harvest planning and was demonstrated to minimize shortfalls in timber volumes over time. Harvest scheduling was simulated by greedy optimization (Cumming and Armstrong 2001) within each time step and was deterministic conditional on the outcome of the fire modules. Scheduling rules and cost equations were based on the existing mixed-tenure (coniferous/deciduous) planning environment of the study region (Cumming and Armstrong 2001; 2004). Timber volumes were calculated from the forest inventory using seven yield class (Table 5-4) described by growth and yield tables developed for ALPAC by Timberline Forest Inventory Consultants Ltd. (Anonymous 1994). The annual allowable cut (AAC) of deciduous and coniferous wood was calculated at the beginning of each time step based on the Hanzlik formula (Hanzlik 1922) which estimates the rate of cut by dividing the volume of mature timber by the average rotation age and adding the average growth rate from under-mature timber. When stands were harvested, all timber volume was removed and the stand was set to age zero in the designated regeneration yield class (Table 5-4). The regeneration rules varied between scenarios based on the current forest management (FM) policy and the alternative regeneration management (AM) policy (Table 5-4).

Regeneration and succession

We simulated regeneration and stand dynamics for all stands with a set of simplified rules based on seven yield classes (Table 5-4). The yield classes simulate differences in the growth and death of trees through time, resulting in

changing composition and wood volume. The trajectory of individual stands within the mixedwood boreal forest is variable, especially on mesic sites where successional trajectories depend strongly on variation in disturbance history, composition, and spatial context (Chen and Popadiouk 2002; McIntire et al. 2005; Johnstone and Chapin 2006a). Tardis simplifies these complex local dynamics using a set of decision rules that are deterministic in the absence of fire. When stands were burned they always regenerated to their pre-burn yield class. When peatlands burned they were regenerated as peatlands; water and anthropogenic development never burned. When stands were harvested, black spruce (yield class 5), jack pine (yield class 6) and aspen/pine (yield class 7) forests were regenerated to their same yield classes in both the FM and AM forest management scenarios (Table 5-4). However harvested mesic forests (yield classes 1 to 4) regenerated according to defined rules that differed between the FM and AM scenarios (Table 5-4). The FM scenarios were based on current management policy (Cumming and Armstrong 2001) that derives regeneration standards from the forest composition at the time of the initial inventory - a single snapshot in time. Based on this policy much mesic forest is reforested as white spruce (yield classes 3 and 4) even though the natural successional trajectory more typically progresses from aspen to white spruce dominance over time.

The alternative regeneration policy implemented in AM scenarios attempted to minimize the area regenerated as pure white spruce while maintaining adequate, harvestable volumes of spruce on the land base. Harvested mesic forests (yield classes 1 to 4) were regenerated to yield classes with either pure aspen (yield class 1) or dominant (leading) aspen with a significant proportion of white spruce (yield class 2). Yield tables provision white spruce growth into these classes through time. The proportional allocation of harvested stands regenerating to the two classes (1 or 2) was based on the coniferous AAC calculated in the current time step. The decision rule was calibrated based on the conifer volume required for sustainable mill operations ($\approx 1\,500\,000\text{ m}^3/\text{year}$ (Cumming and Armstrong 2001). If the harvestable conifer volume dropped below $1\,520\,000\text{ m}^3/\text{year}$ then harvested mesic stands were allocated as leading

aspen (yield class 2), otherwise they were allocated to yield class 1. We propose that yield class 2 acts as white spruce nursery, sustaining substantial growth of the white spruce volume while minimizing fire hazard. In the absence of further disturbance these stands could succeed into a pure white spruce state. The regeneration strategy decision was sub-optimal since the method was not forward-looking; the outcome of regeneration decisions on harvestable white spruce volumes would not be realized for 50 – 100 years.

Scenario analysis

We used three classes of indicators to measure simulated trends in fire, forest harvest and forest composition. The first indicator was the number of fire initiations per time step. The second class included four categories describing regional forest composition including the area: burned in the time step, harvested in the time step, of aspen dominated mesic forest, and of white spruce dominated mesic forest. We refer to initiation and area burned as fire activity indicators. Mesic stands were classified as aspen- or white spruce-dominated based on proportional timber volume; when the volume of white spruce was greater than 50% the stand was classified as white spruce-dominated (Table 5-1). We highlighted the mesic forest type because regional harvest operations alter the structure (composition and age structure) of these forests, there is a strong gradient in fire activity between aspen and white spruce dominated stands, and one of the dominant trajectories of mesic succession is a transition from aspen to white spruce through time. The third indicator group was the burn ratio of aspen, white spruce, jack pine and black spruce forests. Burn ratios provided complementary information to the regional area burned indicator by quantifying the allocation of area burned among forest types.

We calculated a ratio of change (R_A) to synthesize the change in initiation, area burned and burn ratio associated with the treatment factors. Stochasticity generated by the interaction of the fire module and the pseudo-random selection of climate data resulted in a distribution of 100 values per time step for each indicator. These data were approximately symmetrically distributed so we

selected the mean of the medians as a comparative measure such that values would be directly comparable to box plot figures; mean and median are equivalent in symmetrical frequency distributions (Sokal and Rohlf 1995). The R_{Δ} values were calculated from the mean of the medians of the distribution for time steps at the beginning (time steps 1 to 5) and/or end (time step 23 to 27) of the experimental interval. To examine the effect of management across disturbance treatments the R_{Δ} was calculated by dividing the indicator values at the end of the experimental interval for NFM by those for AM and FM (e.g., NFM/FM). To examine the effect of climate we calculated the R_{Δ} by dividing the indicators at the end of the experimental interval by those at the beginning (within disturbance treatments). For climate change scenarios, this interval represented $3 \times \text{CO}_2/1 \times \text{CO}_2$ climate conditions. For NCC scenarios with constant $1 \times \text{CO}_2$ conditions the R_{Δ} represented structural changes under contrasting management scenarios in the absence of climate change.

Climate change experiments were simulations where ongoing NFM, FM or AM management strategies were exposed to increasing fire weather conditions. We did not present scenarios where regions were being transformed from relatively pristine conditions (NFM) to managed forests (FM and AM) *coincident* with climate change. We derived a cross-treatment R_{Δ} for initiation and area burned to estimate of the latter effect by combining information from the NFM and FM/AM scenarios. The cross-treatment R_{Δ} values were calculated using the mean median of initiation and area burned at $1 \times \text{CO}_2$ conditions at the beginning of the NFM and at $3 \times \text{CO}_2$ conditions at the end of the FM and AM scenarios (NFM-FM, NFM-AM).

We calculated the climate-only R_{Δ} between $1 \times \text{CO}_2$ and $3 \times \text{CO}_2$ conditions for initiation and area burned in the absence of vegetation and management feedbacks using general methods described in Krawchuk et al. (in review (Chapter 4)). These estimates reflected the expected change in fire activity solely associated with the DSR-sequence count climate variable. The values presented in Krawchuk et al., (in review (Chapter 4)) included two climate indices

(DSR-sequence count and Seasonal Severity Rating) so were not an appropriate contrast to the outcomes of our simulation experiment.

Results

Fire initiation was constant within NCC scenarios (Figure 5-4 a-c) with an R_A close to unity (Table 5-5: within treatment). Within treatment values are calculated within the same management treatment, the cross-treatment values were calculated across management treatments. The cross-treatment R_A indicated similar initiations in AM, FM and NFM scenarios (Table 5-5, Figure 5-4). Initiation increased in all CC scenarios (Figure 5-4 d-f), with the sharpest increase between time steps six and 21, driven by increases in DSR-sequence count between $1 \times \text{CO}_2$ and $2 \times \text{CO}_2$ conditions (Figure 5-2). The R_A values for initiation under climate change were similar for the within treatment, climate-only and cross-treatment assessments of climate change (Table 5-5) and ranged from 37 to 49 %. At $3 \times \text{CO}_2$ conditions (time step 23 onwards) initiation was slightly lower in the AM scenario than the FM scenario (Figure 5-4 f).

The area burned was constant within all NCC scenarios (Figure 5-5 a-c) resulting in within treatment R_A values close to unity (Table 5-5). The cross-treatment R_A values indicated substantially less area burned in the FM and AM scenarios than NFM (Table 5-5, Figure 5-5 a-c). Area burned increased in all CC scenarios but the absolute area was lowest in the FM and AM scenarios (Figure 5-5 d-f). The R_A suggested the AM scenario experienced the greatest increase in area burned with climate change (Table 5-5). There was little difference among the within treatment R_A values for the CC-NFM, CC-FM and the climate-only treatment but a substantial reduction in R_A for the cross-treatment $3 \times \text{CO}_2 / 1 \times \text{CO}_2$ indicators (Table 5-5). The area burned at $3 \times \text{CO}_2$ conditions under FM and AM (Figure 5-5 e,f) was only slightly higher than that burned under NFM at $1 \times \text{CO}_2$ conditions (Figure 5a). The changes in area burned by climate change and forest management on the base, NFM condition were equivalent. The within treatment R_A for CC-NFM and the cross-treatment R_A for NCC-NFM / NCC-FM were both 1.49 (Table 5-5).

The area harvested was consistent in the FM scenarios (Figure 5-5b,e). The FM and AM scenarios had equal areas of forest harvested at the beginning of the experimental interval but the area decreased through time in the AM scenarios (Figure 5-5) reflecting a gradual decrease in the deciduous AAC. The cumulative area disturbed by fire and harvest in AM and FM scenarios was similar, and slightly less than the area burned in the NFM scenarios (Figure 5-6).

There was greater area of white spruce dominated forest in the NFM scenarios than FM or AM, under NCC and CC (Figure 5-7). The area of white spruce forest decreased in the CC-NFM in comparison to the NCC-NFM (Figures 5-7a,d). The area of white spruce forest was relatively stable in the FM scenarios and increased in the AM scenarios; the increase was greatest in the NCC-AM scenario (Figure 5-7c). Aspen dominated and recently burned forest reduce fire size, and the area of these two cover types was greater in the FM and AM scenarios in comparison to NFM (Figure 5-7). The area of aspen dominated and recently burned forest decreased in the NCC-AM and was relatively stable in the CC-AM (Figure 5-7c,f). In comparison, their area was stable in the NCC-FM, and increased in CC-FM (Figure 5-7b,e).

Forest management altered the age structure of aspen dominated and white spruce dominated forests (Figure 5-8). There was more old (≥ 90 years) aspen in the NFM scenarios than the FM or AM, though the area of aspen dominated forest in the AM was more similar to the NFM than the FM scenario (Figure 5-8, top panels). Similarly, there was more old (≥ 120 years) white spruce in the NFM and AM scenarios than the FM (Figure 5-8, bottom panels).

The extension of CC scenarios for 140 years under constant $3 \times \text{CO}_2$ conditions (time steps 28 to 50) yielded relatively constant fire activity in all treatments (Figures 5-4 and 5-5). At the end of the 250 year experiment the areas of old aspen dominated and old white spruce dominated forest in the AM was roughly equivalent to that in the NFM (Figures 5-8 a and c) and there was much less in the FM scenario (Figure 5-8b).

The burn ratios differed among forest cover types and the relative rank was invariant among scenarios; aspen dominated forest always had the lowest

burn ratio (Table 5-6). The absolute value of burn ratios decreased in FM and AM scenarios in accordance with a decrease in area burned relative to the NFM scenario, and all ratios increased under climate change (Table 5-6). The burn ratios of aspen dominated and white spruce dominated mesic forest were similar in AM and FM scenarios, though the burn ratio for white spruce was smallest in the NCC-AM scenario (Table 5-6). The R_A between $3 \times \text{CO}_2$ and $1 \times \text{CO}_2$ conditions was similar among all forest types, but the largest R_A occurred in the AM scenario.

Discussion

Climate change increased lightning fire initiation and area burned, and altered regional forest composition. Young, relatively fire-resistant aspen dominated forests became more abundant as area burned increased by 49 to 83 % between $1 \times \text{CO}_2$ and $3 \times \text{CO}_2$ conditions but the magnitude of this increase was insufficient to generate regional, negative feedback to subsequent activity. The ratios of change for both initiation and area burned calculated from our simulations were equivalent to those from climate data alone. However, the negative fire feedback hypothesis should not be dismissed outright. There is considerable uncertainty in predictions made by climate change models (Allen et al. 2000; Rupp et al. 2007) and different indices derived from the same model can lead to varying expectations in future climate (Flannigan et al. 1998) such that the rates of climate change used in this experiment might under- or over-predict net changes in future fire weather. Fire initiation was the only process that responded to climate change in our simulations, and the century long interval of climate increase (CRCM data from 1975 to 2089) was relatively short compared to the longevity of boreal trees and thus forest dynamics. A greater change in fire activity over an extended experimental interval, or incorporating changes in post-disturbance vegetation community structure generated *via* the interaction of climate change and productivity (e.g., Rupp et al. 2002; Scheller and Mladenoff 2005; Hogg and Bernier 2005; Schumacher and Bugmann 2006) might still generate feedback dynamics. In future work, potential changes in the

effectiveness of fire suppression under climate change should also be taken into account more fully (Flannigan et al. 2005) via changes in the probability of escape (Cumming 2005; Arienti et al. 2006) or fire size (Cumming 2001b). Wotton and Flannigan (1993) suggest that the fire season might begin earlier in our study region under $2 \times \text{CO}_2$ relative to $1 \times \text{CO}_2$ conditions (May 16 versus May 30). Our time series of climate data was generated from fire weather projections from 1 May to 30 September such that the extension of the fire season into the spring was included in our simulations as would further extensions expected under $3 \times \text{CO}_2$ conditions. Our data did not account for a nine day projected extension of the fire season into the fall. An extended season would allow for existing fires to burn longer, further changing the fire size distribution under future climate conditions.

Forest harvest caused changes in landscape composition that systematically reduced fire activity under constant climate and climate change conditions. As such, though we found no evidence for self-regulating feedback in fire (above), climate change scenarios omitting anthropogenic development will certainly present biased projections of the future. Our *post hoc* cross-treatment ratios of change suggested the onset of forest harvest coincident with increases in fire weather substantially decreased the change in area burned expected in the future, reducing it to between 22 to 29% from the 61% expected from climate change alone. Ongoing forest harvest in Canada's boreal forests over the past decades (Kasischke and Turetsky 2006) might be masking some of the effects of increasing fire weather conditions on fire activity. Westerling et al. (2006) also emphasize the need to partition the effects of climate change and human-caused changes in land cover in their study of climate-induced changes in fire activity in the western United States. Hirsch et al. (2001) demonstrated a reduction in area burned within a western Alberta landscape could be achieved via forest harvest through the specific configuration of harvest blocks. We demonstrate that the amount of forest harvest occurring under both current and alternative management strategies decreases fire activity regardless of patch configuration, and further quantify the systematic effect of forest harvest in reducing the fire activity expected with climate change. Fire initiation was sensitive to the area of recently

harvested forest in our model, based on previous work that suggested an increased probability of fire initiation in these areas (Krawchuk and Cumming submitted (Chapter 3)). Future work should explore the effect of recently harvested area on fire size. Fire operations staff in the region consider harvest slash, in combination with grasses that quickly invade harvest blocks, as high fuel hazards (Hirsch et al. 2001; Baxter 2004; Kurt Frederick, Alberta Sustainable Resource Development, pers. comm.). Under these conditions the magnitude of the reduction in area burned observed in our FM and AM scenarios might be less, however the amount of young, deciduous dominated mesic forest generated through forest harvest will still act as a strong buffer to fire activity.

We presented an alternative management scenario as an adaptive strategy to reduce fire activity. There was limited change in the area burned between the harvest (FM and AM) scenarios, especially in contrast to the significant reduction in area burned shown by harvested scenarios relative to the non-harvest (NFM) scenario. More substantial changes in regeneration alternatives or the amount and configuration of harvest activities will be required to reduce fire activity beyond the already large effect that current rates of forest harvest generate, regardless of the selection of a particular regeneration strategy. Although there was limited effect on area burned, the alternative regeneration policy had a considerable effect on the age structure of mesic forests. The age distribution differed substantially between the AM and FM scenarios. There was less young white spruce forest in the AM scenarios, as expected. But there was more old aspen and white spruce forest such that there were equivalent areas of white spruce forest overall in the AM and FM scenarios at $3 \times \text{CO}_2$ conditions. One might infer this pattern could be generated by a longer fire return interval in mesic forests in the AM scenarios; since the areas burned in the AM and FM scenarios were equivalent the non-mesic forest types would need to more frequently 'burn up the slack'. However the burn ratios for all forest types were very similar in AM and FM scenarios, the burn ratio for white spruce was only marginally lower in the AM than the FM under $1 \times \text{CO}_2$ climate conditions. Finer scaled simulations may be needed to

explore the specific dynamics generating the simulated variation in forest age structure between the current and alternative management scenarios.

Forest yield and harvest was relatively constant in the FM scenarios however the area and volume harvested decreased slightly through the AM scenarios. A closer inspection of the AAC output showed that conifer volumes increased systematically and deciduous volumes decreased in the AM scenarios. This behaviour suggests the switch value for regeneration decision rules in the AM scenarios requires further refinement. The threshold value used as a switch for the regeneration sequence within the AM scenarios was less than optimal because the model did not have the capacity to look forward in time and assess future wood volumes in the region; it was reactive rather than proactive. The scenario favoured the regeneration of white spruce when the coniferous AAC fell below $1\,520\,000\text{ m}^3/\text{year}$. However the time lapse between this regeneration switch and the actual harvest of white spruce is ~ 100 years. The difference in harvested area between the AM and FM scenarios could be contributing to the contrasting age structure in mesic stands, however the effect would be minimal since decreases in harvested area were relatively small and began towards the end of the simulation interval. The simulation interval of 250 years represented less than three management rotations after the burn-in period of 155 years. Due to the long rotation ages, a longer simulation period provided by an increased extent in climate projections would be beneficial, providing a broader assessment of forest system behaviour. We did not examine the direct influence of climate change on mill operations, an assessment of the long-term economic stability of forestry operations are beyond the focus of this study.

Forest management decreased regional fire activity but the forest age structure that remained after harvesting was shifted substantially from natural conditions (NFM scenarios). This shift illustrates a trade-off between the minimization of fire activity and the emulation of natural disturbance. The removal of aspen and white spruce forest from mesic stands was a primary focus of harvest operations. In contrast, fire preferentially burns conifers (black and white spruce, jack pine) over aspen. Cumming (2001a) commented on the

differential selection by wildfire and harvest and underlined that this difference alone debases the potential for the emulation of natural disturbance by forest harvest activity. The area burned in NFM was equivalent to the area burned and harvested in AM/FM scenarios but the effective differences from the selection by harvest and fire were evident in the comparison of forest composition between NFM and AM/FM scenarios. The area of old aspen increased in the CC-NFM illustrating that while fire activity increased with climate change the resistance of aspen forest to fire supported the development of old growth stands. In contrast, the area of old aspen forest was reduced by harvesting in the FM, but less so in the AM scenarios. The area of white spruce dominated mesic forest was reduced in all harvesting scenarios. The AM scenario had more old aspen and white spruce than the FM scenario, but (as expected) less young white spruce. Though the AM policy did not reduce fire activity substantially relative to the FM policy, it generated a mesic forest structure more similar to NFM than FM. This alternative policy may be a better mechanism for ecosystem management, especially for old growth specialists. It would certainly provide an interesting treatment for an adaptive management experiment. The non-equilibrium fluctuations in NFM forest composition is a stark contrast to the more static conditions generated with the FM scenario. The fluctuations are consistent with studies suggesting that there are no representative areas in the mixedwood boreal due to the continuing cycles in forest composition over very broad temporal and spatial scales (Cumming et al. 1996).

Our simulation experiment suggests that forest harvest and climate change have opposing effects on the fire regime in Alberta's mixedwood boreal forest. Climate change increases fire activity, however the magnitude of the resulting increase in fire initiation and area burned was not sufficient to cause a negative feedback between recently burned areas and subsequent fire activity. Regardless of changes in climate, the area burned by lightning initiated fires decreased as a function of forestry (either FM or AM policies) in comparison to those with no harvest activity due to changes in forest composition, e.g., the area of aspen dominated forest on the landscape. Even so, fire behaviour may be exacerbated in

some harvested landscapes as a function of increases in fires from human activities (Arienti et al. 2006; Arienti et al. in review), extreme fire weather conditions that may burn through aspen dominated stands (Quintilio et al. 1991) or/and large areas of recently harvested forest area providing large extents of fine fuels (e.g., Baxter 2003; Krawchuk and Cumming in prep.(Chapter 3)). Forest harvesting may act as a systematic buffer against climate change expected in the future however the resulting decrease in fire activity is accompanied by a marked transition in forest age structure relative to un-harvested conditions. The decrease in fire activity afforded by forest harvest comes at a cost of the selective removal of forest types and ages atypical under the natural disturbance regime. This discrepancy could affect those species, communities and processes that rely on forest types such as old mesic stands. Boreal species have adapted to fire and the conditions it generates over millenia (Weber and Stocks 1998) and this long fire history has not been static. Climate shifts as recent as the Little Ice Age hosted interrelated changes in the composition of vegetative communities and fire regime (Weber and Stocks 1998; Weir et al. 2000). However the rate and interaction of a changing climate with land cover fragmentation and conversion might outpace nature's ability to adapt (Noss 2001).

Resilience theory suggests that disturbance beyond the natural range of variation can trigger transition to an alternative stable state (Holling 1973; Scheffer et al. 2001). In previous work (Krawchuk and Cumming submitted. (Chapter 3)), we demonstrate that the local spatial interaction of forest harvest and wildfire could initiate compound disturbances. We suggest that multiple events could result in a transition to a new ecological regime of aspen parkland, reinforced by changes in vegetation communities caused by climate change (Hogg and Bernier 2005). Hogg and Hurdle (1995) illustrate that climate change within the century could shift isolines of the climate moisture index that have been shown to reflect the boundary of the mixedwood forest. This shift could initiate a transition from mixedwood boreal forest to aspen parkland (Hogg and Hurdle 1995). The aspen parkland is a vegetation community dominated by grasses (*Calamagrostis canadensis* Michx.) and low densities of aspen, which could

induce further changes in fire activity. Drought induced post-fire regeneration failure and mortality of existing trees has already been observed in mixedwood forests in the Yukon and Alberta (Hogg et al. 2005; Hogg and Wein 2005). Future development of the Tardis model should include climate-vegetation feedbacks into post-fire and post-harvest regeneration modules that would further our understanding of future boreal forest development. Payette and Delwaide (2003) show that compound disturbances of forest harvest, fire and insect outbreaks have resulted in state transition from boreal spruce to lichen heathland in the near term, and Jasinski and Payette (2005) show a long-term transition in stable state from spruce-moss forest to spruce-lichen woodland through the interaction of insect activity and wildfire in the same region. Within the last year, the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) has extended its outbreak range from British Columbia into western Alberta. As this beetle continues to extend its outbreak range into eastern Alberta this additional, biotic disturbance will add further complexity to the disturbance-climate-vegetation feedbacks that currently give rise to the boreal mixedwood forest. The interaction of climate change, forest management, wildfire and insect outbreaks may be sufficient to prompt changes from boreal mixedwood forest to aspen parkland more quickly than might occur from any of these factors in isolation.

Table 5-1. Landscape composition of the study region at the beginning of the simulations.

Cover type	Description	Percent
old aspen mesic forest	mesic forest with up to 50% aspen by volume; ≥ 90 years	1
young aspen mesic forest	mesic forest with up to 50% aspen by volume; < 90 years	26
young white spruce mesic forest	mesic forest with up to 50% white spruce by volume; ≤ 120 years	2
old white spruce mesic forest	mesic forest with up to 50% white spruce by volume; > 120 years	4
jack pine forest	jack pine dominated stands	9
black spruce forest	black spruce dominated stands	10
muskeg or open scrub	low density scrub and all wetland types (not including water bodies)	37
water	permanent water bodies	3
burned	burned within 30 years	7
harvested	harvested within 30 years	0.7
anthropogenic development	permanent human development	0.3

Table 5-2. Covariates and estimated parameters of the zero-inflated negative binomial model of lightning fire initiation per landscape per year. In the binomial component estimates are for $P(0|X)$ rather than $P(1|X)$. Goodness of fit was estimated using a parametric bootstrap (Efron and Tibshirani 1998) of the sums of squares ($SS = 1042$, $n = 1000$, $p=0.28$). The ΔAIC describes the change in Akaike's Information Criterion with the removal of each covariate from the model.

Covariate	Estimate (s.e.)	ΔAIC
intercept (binomial)	+1.7 (0.2)	
DSR sequence count	-0.4 (0.05)	192
intercept (negative binomial)	-1.4 (0.3)	
area harvested within 10 years	+6.0 (1.5)	12
area of black spruce dominated forest > 30 years	+2.2 (1.03)	4
squared area of black spruce dominated forest > 30 years	-2.4 (1.06)	2
area of white spruce dominated mesic forest > 30 years	+2.6 (0.8)	8
area of deciduous dominated mesic forest >30 and <90 years	-2.4 (0.5)	24
area burned within 30 years	-1.3 (0.4)	4
theta	+3.16 (1.4)	6

Table 5-3. Protocol for the restricted, random sampling of the DSR-sequence count indices for time steps in Tardis simulations. The indices were generated from Canadian Regional Climate Model (CRCM) data (Laprise et al. 1998). Each time step was five years so five restricted-random selections were drawn from the annual climate data.

Simulation time-step	CRCM climate data	Julian years
burn-in (31 time steps)	1 $\times$ CO ₂	
1 - 5	5 @ 1 $\times$ CO ₂	1975 - 1985
6 - 7	4 @ 1 $\times$ CO ₂ , 1 @ 2 $\times$ CO ₂	
8 - 9	3 @ 1 $\times$ CO ₂ , 2 @ 2 $\times$ CO ₂	
10 - 11	2 @ 1 $\times$ CO ₂ , 3 @ 2 $\times$ CO ₂	
12-13	1 @ 1 $\times$ CO ₂ , 4 @ 2 $\times$ CO ₂	
14 - 16	5 @ 2 $\times$ CO ₂	2040 - 2049
17 - 18	4 @ 2 $\times$ CO ₂ , 1 @ 3 $\times$ CO ₂	
19 - 20	3 @ 2 $\times$ CO ₂ , 2 @ 3 $\times$ CO ₂	
21	2 @ 2 $\times$ CO ₂ , 3 @ 3 $\times$ CO ₂	
22	1 @ 2 $\times$ CO ₂ , 4 @ 3 $\times$ CO ₂	
23 - 24	5 @ 3 $\times$ CO ₂	2080 - 2089
25 - 50	5 @ 3 $\times$ CO ₂	2090 onwards

Table 5-4. The regeneration rules for harvested stands in current forest management (FM) and alternative forest management (AM) scenarios. The proportion of the forested area in each yield class at the beginning of the simulation is listed under 'Area'.

Pre-harvest yield class	FM	AM	Area
1 (deciduous dominated mesic)	1	1 or 2*	0.28
2 (deciduous dominated mesic with significant sub-dominance by white spruce)	2	2	0.14
3 (white spruce dominated mesic with significant sub-dominance by deciduous)	3	2	0.04
4 (white spruce dominated mesic)	4	2	0.08
5 (black spruce dominated non-mesic)	5	5	0.28
6 (jack pine dominated non-mesic)	6	6	0.14
7 (deciduous dominated non-mesic with significant sub-dominance by jack pine)	7	7	0.04

*depending on the volume (m^3) of coniferous forest in the region. See text for details.

Table 5-5. R_A values for initiation and area burned. For within treatment contrasts: NCC (no climate change) values were calculated using time steps 23 through 27 versus 1 through 5, representing $1 \times \text{CO}_2/1 \times \text{CO}_2$; CC (climate change) values were calculated over the same time steps representing $3 \times \text{CO}_2/1 \times \text{CO}_2$. Cross-treatment contrasts are unique and given in the text. The three management treatments are: no forest management (NFM), current forest management (FM) and alternative forest management (AM).

Contrasts	Scenario	Initiation	Area burned
Within treatment	NCC-NFM ($1 \times \text{CO}_2/1 \times \text{CO}_2$)	0.94	0.97
	NCC-FM ($1 \times \text{CO}_2/1 \times \text{CO}_2$)	1.00	0.95
	NCC-AM ($1 \times \text{CO}_2/1 \times \text{CO}_2$)	1.02	1.03
	CC-NFM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.37	1.49
	CC-FM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.39	1.47
	CC-AM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.49	1.83
	climate-only ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.48	1.61
Cross-treatment	NCC-NFM / NCC-AM ($1 \times \text{CO}_2/1 \times \text{CO}_2$)	1.023	1.47
	NCC-NFM / NCC-FM ($1 \times \text{CO}_2/1 \times \text{CO}_2$)	0.98	1.49
	CC-NFM / CC-AM ($3 \times \text{CO}_2/3 \times \text{CO}_2$)	1.0084	1.17
	CC-NFM / CC-FM ($3 \times \text{CO}_2/3 \times \text{CO}_2$)	0.99	1.26
	CC-FM / NCC-NFM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.44	1.22
	CC-AM / NCC-NFM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.42	1.29

Table 5-6: Burn ratios calculated from time steps 23 through 27 for each of six scenarios (rows 1 - 6). The ratios of change (R_{Δ}) of these burn ratios from climate change (CC) scenarios are provided in rows 6 - 9. The three management treatments are: no forest management (NFM), current forest management (FM) and alternative forest management (AM).

Scenario	Aspen	White spruce	Black spruce	Jack pine
NCC-NFM ($1 \times \text{CO}_2$)	0.017	0.064	0.14	0.26
NCC-FM ($1 \times \text{CO}_2$)	0.013	0.043	0.086	0.19
NCC-AM ($1 \times \text{CO}_2$)	0.013	0.036	0.092	0.20
CC-NFM ($3 \times \text{CO}_2$)	0.028	0.10	0.22	0.41
CC-FM ($3 \times \text{CO}_2$)	0.023	0.076	0.16	0.35
CC-AM ($3 \times \text{CO}_2$)	0.022	0.076	0.17	0.34
R_{Δ} CC-NFM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.41	1.52	1.34	1.36
R_{Δ} CC-FM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.64	1.46	1.49	1.53
R_{Δ} CC-AM ($3 \times \text{CO}_2/1 \times \text{CO}_2$)	1.85	1.64	1.95	1.85

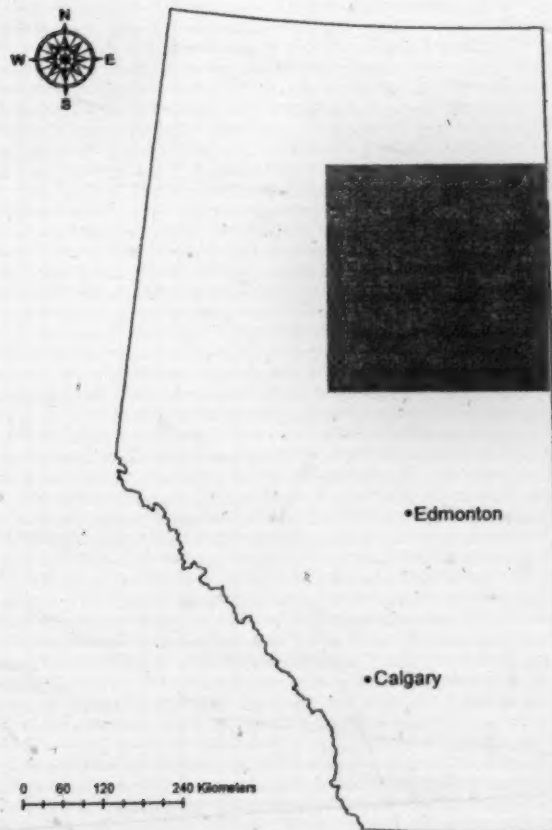


Figure 5-1. The study region is illustrated in grey, within Province of Alberta, Canada.

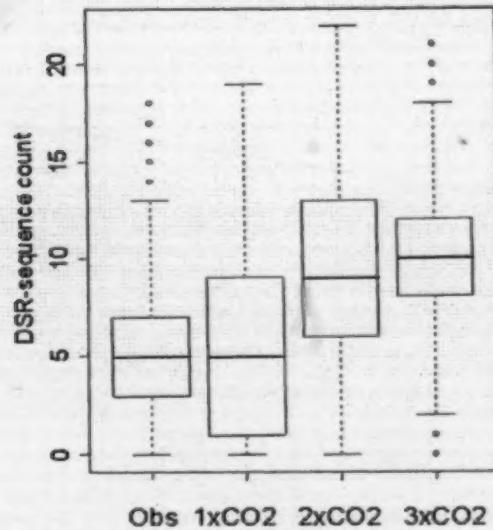
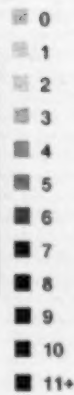


Figure 5-2. The distribution in annual DSR-sequence count (from Krawchuk et al. in review; Chapter 4) for observed (Obs) and simulated $1 \times \text{CO}_2$, $2 \times \text{CO}_2$ and $3 \times \text{CO}_2$ conditions from the Canadian Regional Climate Model (Laprise et al. 1998). The 'Obs' distribution is composed of annual, empirical data from conditions observed between 1983 and 2001.

a) Initiations



b) Area burned

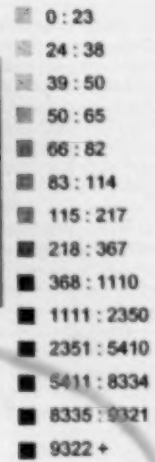
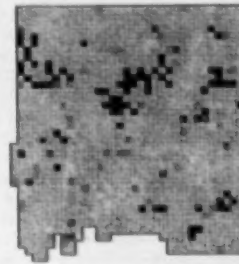
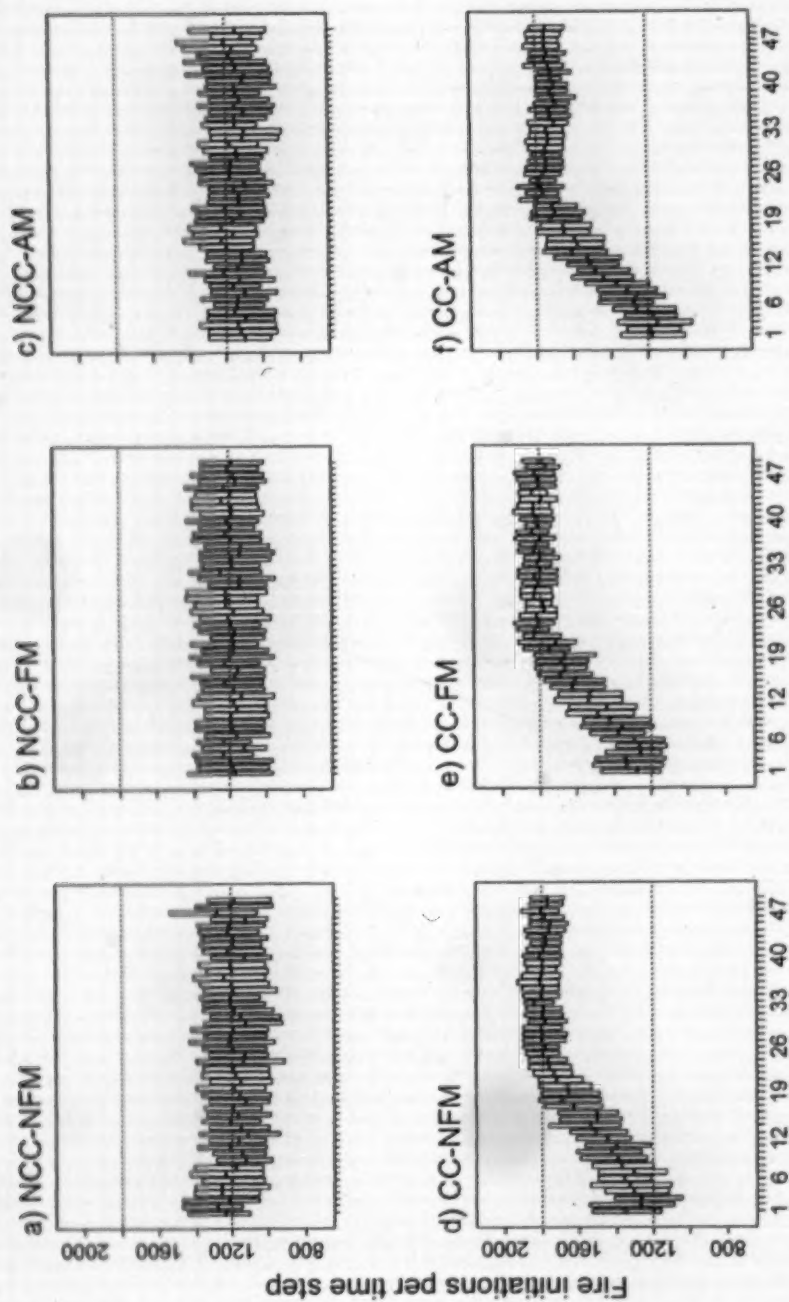


Figure 5-3. A sample of spatial patterns in a) initiation (initiations per time step) and b) area burned (hectares per time step) generated by the Tardis simulation model.



Time steps

Figure 5-4. The interquartile distributions of lightning fire initiation under constant climate (NCC; a,b,c) and climate change (CC; d,e,f) conditions for three management scenarios: no forest management (NFM), current forest management (FM) and alternative forest management (AM). The dashed lines are for visual reference.

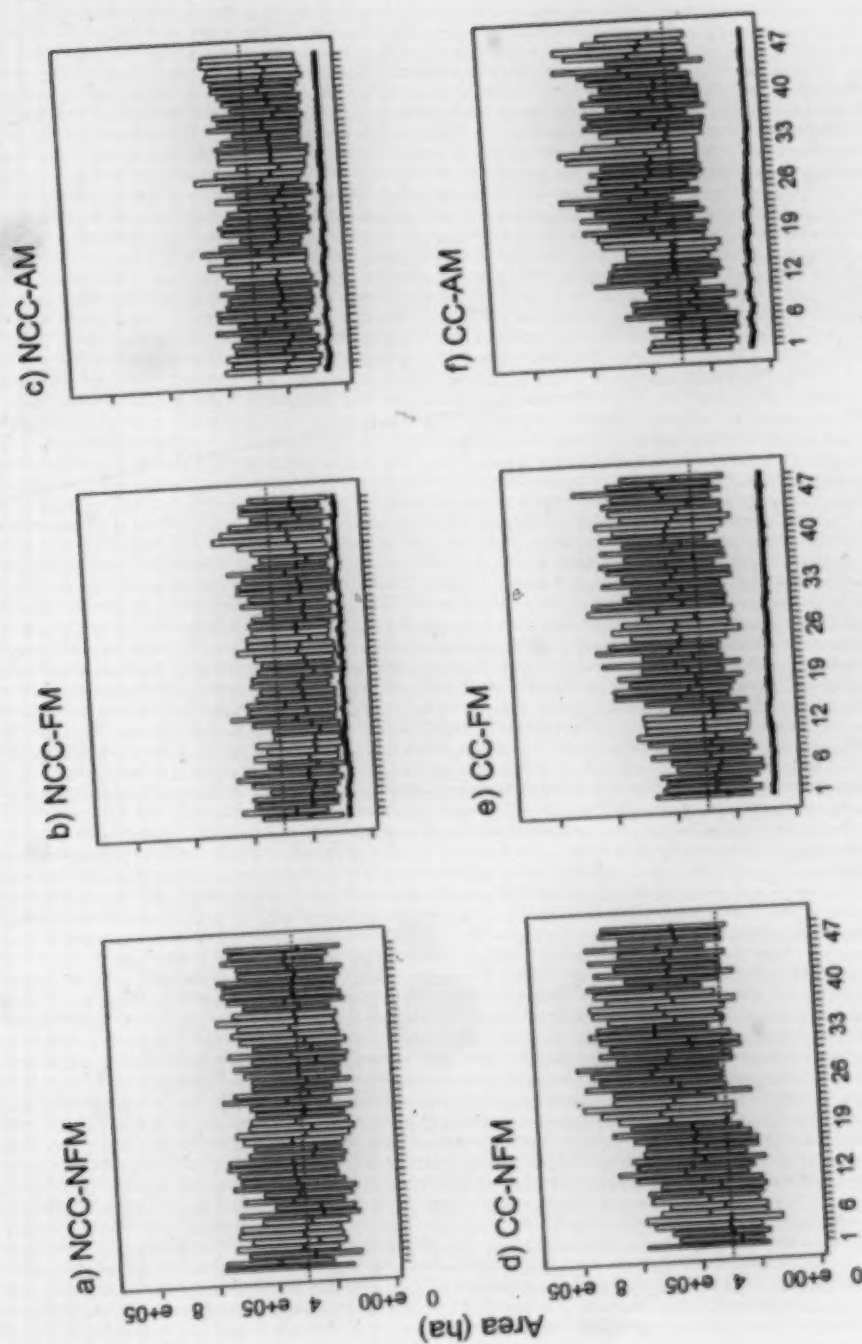


Figure 5-5. The interquartile distributions of area burned (white) and harvested (black) under constant climate (NCC; a,b,c) and climate change (CC; d,e,f) conditions for three management scenarios: no forest management (NFM), current forest management (FM) and alternative forest management (AM). The dashed line marks 298 673 ha, the grand median area burned in NCC-NFM (an annual burn rate of 0.6%).

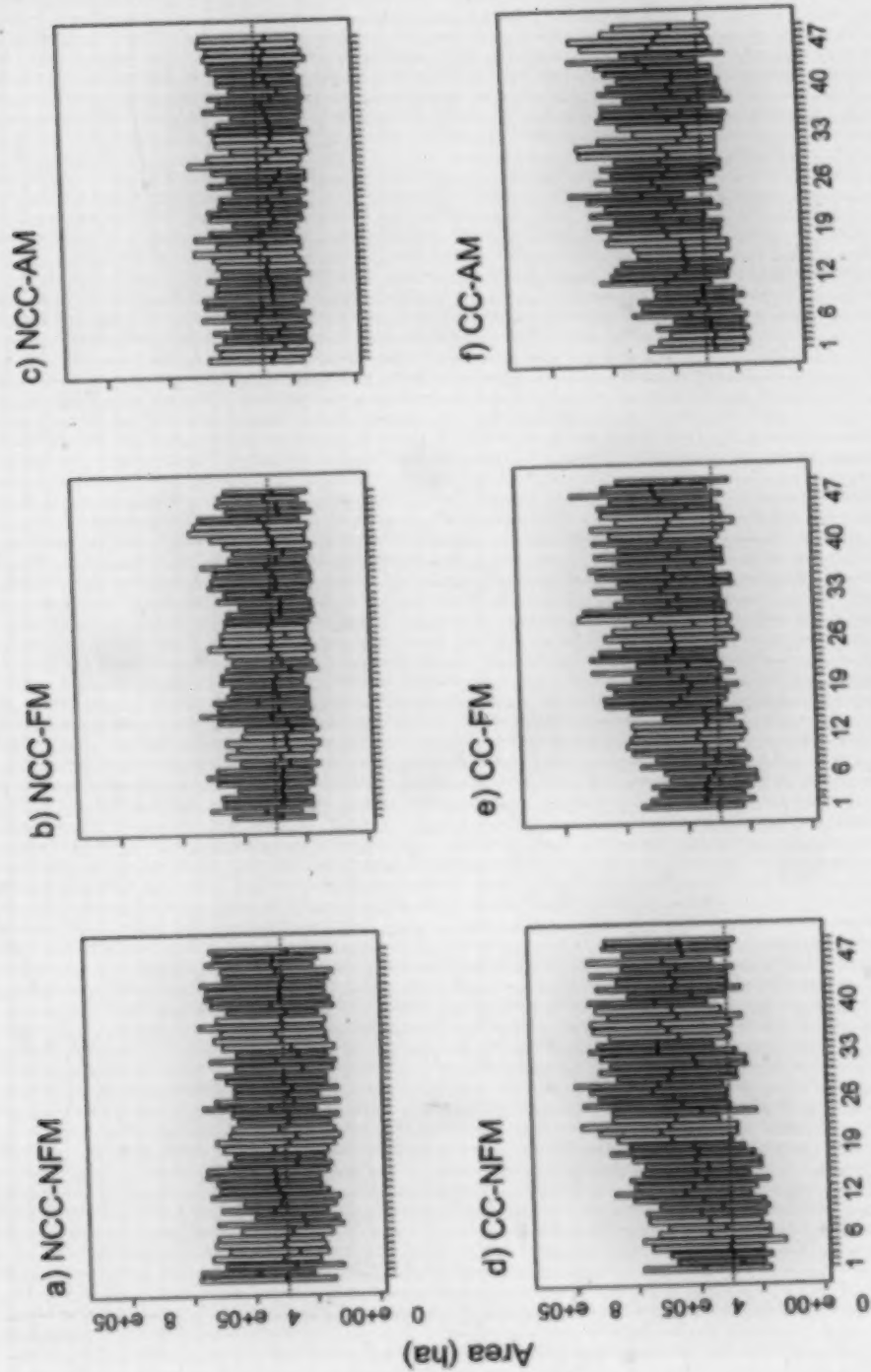
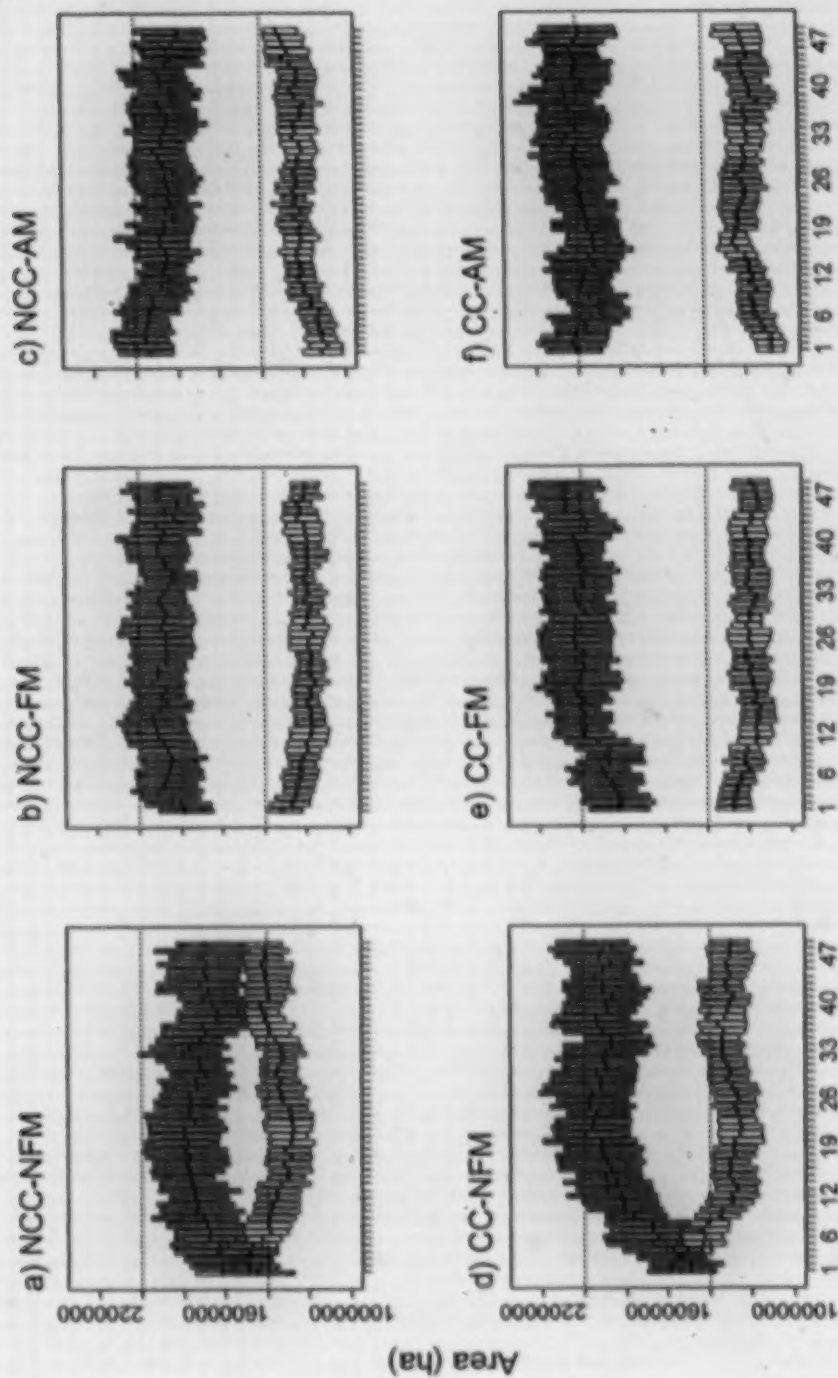


Figure 5-6. The interquartile distributions of area disturbed (burned + harvest) under constant climate (NCC; a,b,c) and climate change (CC; d,e,f) conditions for three management scenarios: no forest management (NFM), current forest management (FM) and alternative forest management (AM). The dashed lines marks 298 673 ha, the grand median area burned in NCC-NFM.



Time step

Figure 5-7. The interquartile distributions of the area of white spruce dominated forest (white) and the cumulative area of recently burned and aspen dominated forest (grey) under constant climate (NCC; a,b,c) and climate change (CC; d,e,f) conditions for three management scenarios: no forest management (NFM), current forest management (FM) and alternative forest management (AM). The dashed lines are for visual reference.

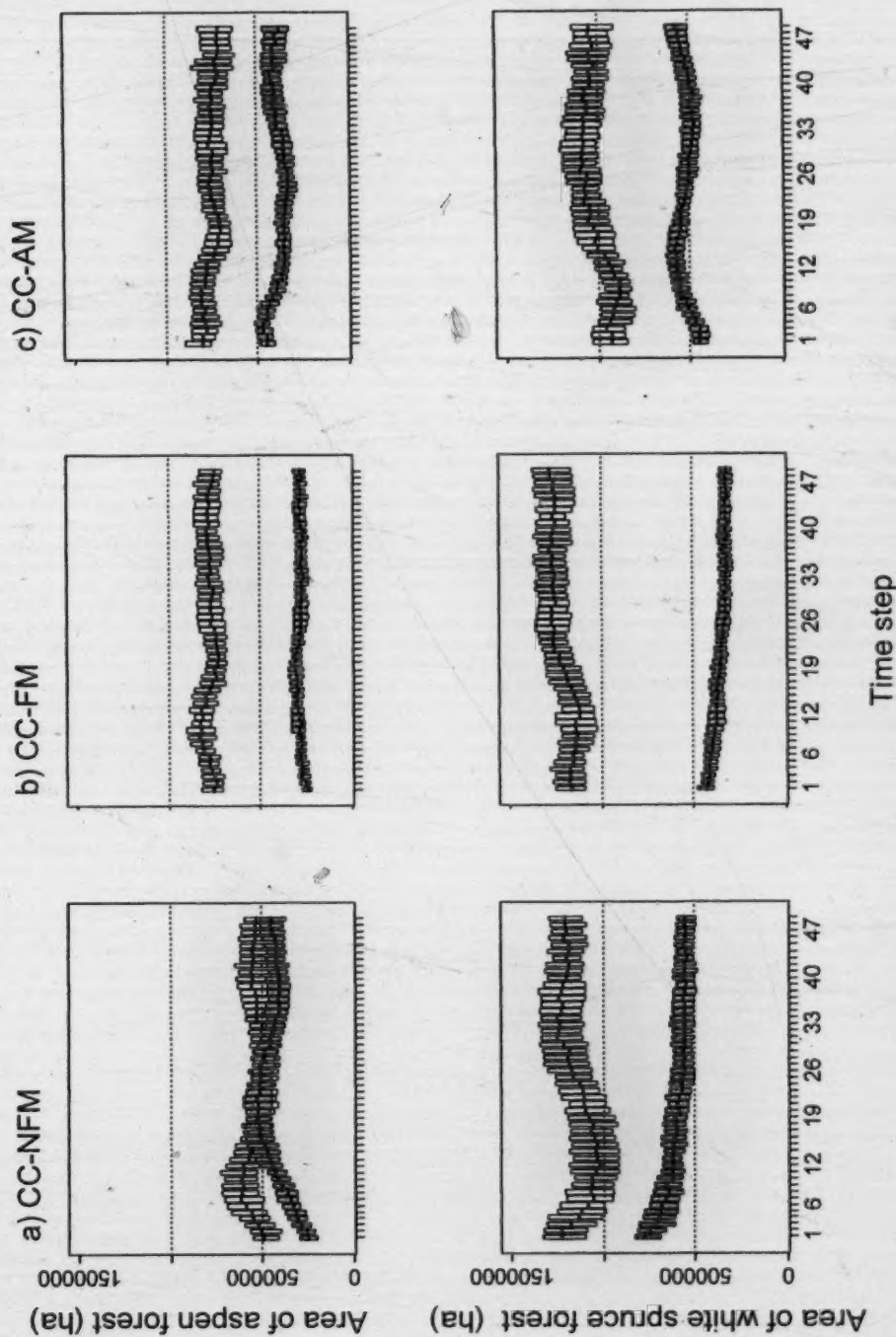


Figure 5-8. The interquartile distributions of the area of young (white) and old (grey) aspen dominated mesic forest (top panels) and white spruce dominated mesic forest (bottom panels) in climate change (CC) scenarios for three management scenarios: no forest management (NFM), current forest management (FM) and alternative forest management (AM). The dashed lines are for visual reference.

Literature Citations

- Albani, M., D.W. Andison and J.P. Kimmins. 2005. Boreal mixedwood species composition in relationship to topography and white spruce seed dispersal constraint. *Forest Ecology and Management* **209**:167-180.
- Alberta-Pacific Forest Industries, Inc. 2004. Alberta-Pacific FMA Area Forest Management Plan September 2004. Boyle, Alberta, Canada.
- Alberta Sustainable Resources and Development. 2006.
http://www.srd.gov.ab.ca/wildfires/fpd/wi_hdtys_10_yr_average.cfm;
accessed 9 June, 2006.
- Alexander, M.E. 1982. Fire behavior in aspen slash fuels as related to the Canadian Fire Weather Index. *Canadian Journal of Forest Research* **12**:1028-1029.
- Allen, M. R., P.A. Stott, J.F.B. Mitchell, R. Schnur and T.L. Delworth. 2000. Quantifying the uncertainty in forecasts of anthropogenic climate change. *Nature* **407**:617-620.
- Anonymous. 1985. Alberta Phase Three forest inventory: forest cover type specifications. Alberta Forestry, Lands and Wildlife, Forest Service ENR Report Department 58.
- Anonymous. 1994. Stand projection yield curves and yield classes for the mid boreal mixedwood ecoregion and the upper boreal mixedwood ecoregion. Technical report, Pearson Timberline Forestry Consultants, Edmonton, Alberta.
- Arienti, M.C., S.G. Cumming and S. Boutin. 2006. Empirical models of forest fire initial attack success probabilities: the effects of fuels, anthropogenic linear features, fire weather, and management. *Canadian Journal of Forest Research* **36**:3155-3166.
- Armstrong, G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. *Canadian Journal of Forest Research* **29**:424-433.
- Baxter, G. 2003. Issues arising from observed fire behaviour in aspen cutblock during the House River Fire in Alberta May-June 2002: A case study. Forest Engineering Research Institute of Canada.
- Baxter, G. 2004. Managing of harvest debris in east-central Alberta. *Forest Engineering Research Institute of Canada, Advantage* **5**:1-10.
- Beaubien, E.G. and H.J. Freeland. 2000. Spring phenology trends in Alberta,

- Canada: links to ocean temperature. *International Journal of Biometeorology* **44**:53-59.
- Bessie, W.C. and E.A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forest. *Ecology* **76**:747-762.
- Bonan, G. B. and H.H. Shugart. 1989. Environmental factors and ecological processes in boreal forests. *Annual Review of Ecology and Systematics* **20**:1-28.
- Chapin, F.S., A.L. Lovcraft, E.S. Zavaleta, J. Nelson, M.D. Robards, G.P. Kofinas, S.F. Trainor, G.D. Peterson, H.P. Huntington and R.L. Naylor. 2006. Policy strategies to address sustainability of Alaskan boreal forests in response to a directionally changing climate. *Proceedings of the National Academy of Sciences of the United States of America* **103**:16637-16643.
- Chapin, F.S., A.D. McGuire, J. Randerson, R. Pielke, D. Baldocchi, S.E. Hobbie, N. Roulet, W. Eugster, E. Kasischke, E.B. Rastetter, S.A. Zimov and S.W. Running. 2000. Arctic and boreal ecosystems of western North America as components of the climate system. *Global Change Biology* **6**:211-223.
Notes: Supplement: Suppl. 1
- Chen, H.Y.H. and R.V. Popadiouk. 2002. Dynamics of North American boreal mixedwoods. *Environmental Reviews* **10**:137-166.
- Clark, J.S. 1998. Why trees migrate so fast: confronting theory with dispersal biology and the Paleorecord. *American Naturalist* **152**:204-224.
- Cumming, S.G. 1997. Landscape dynamics of the boreal mixedwood forest. Ph.D. Dissertation. University of British Columbia, Vancouver, B.C., Canada.
- Cumming, S.G. 2001a. Forest type and wildfire in the Alberta boreal mixedwood: what do fires burn? *Ecological Applications* **11**:97-110.
- Cumming, S.G. 2001b. A parametric model of the fire-size distribution. *Canadian Journal of Forest Research* **31**:1297-1303.
- Cumming, S.G. 2005. Effective fire suppression in boreal forests. *Canadian Journal of Forest Research* **35**:772-786.
- Cumming, S.G. and G.W. Armstrong. 2001. Divided land base and overlapping forest tenure in Alberta, Canada: a simulation study exploring costs of forest policy. *Forestry Chronicle* **77**:501-508.
- Cumming, S.G. and P. Vernier. 2002. Statistical models of landscape pattern metrics, with applications to regional scale dynamic forest simulations. *Landscape Ecology* **17**:433-444.

- Cumming, S.G. and G.W. Armstrong. 2004. Divided landbase, overlapping tenures, and fire risk. *Forestry Chronicle* 80:478-484.
- Cumming, S.G., Burton, P.J. and B. Klinkenberg. 1996. Boreal mixedwood forests may have no "representative" areas: some implications for reserve design. *Ecography* 19:162-180.
- Delisle, G.P. and R.J. Hall. 1987. Forest fire history maps of Alberta, 1931 – 1983. Canadian Forest Service, Northern Forestry Centre, Edmonton Alberta.
- Efron, B. and R.J. Tibshirani. 1998. *An Introduction to the Bootstrap*. Chapman and Hall/CRC, New York.
- Flannigan, M.D., Y. Bergeron, O. Engelmark and B.M. Wotton. 1998. Future wildfire in circumboreal forests in relation to global warming. *Journal of Vegetation Science* 9:469-476.
- Flannigan, M.D. and J.B. Harrington. 1988. A study of the relation of meteorological variables to monthly provincial area burned by wildfire in Canada (1953-80). *Journal of Applied Meteorology* 27:441-452.
- Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner and B.J. Stocks. 2005. Future area burned in Canada. *Climatic Change* 72:1-16.
- Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian Forest Fire Behavior Prediction System. Forestry Canada, Ottawa.
- Gardner, R.H., W.W. Hargrove, M.G. Turner and W.H. Romme. 1996. Climate change, disturbances and landscape dynamics. In: Walter, B. and W. Steffen (eds.), *Global change and terrestrial ecosystems*. Cambridge University Press, pp. 149-172.
- Gillett, N.P., A.J. Weaver, F.W. Zwiers and M.D. Flannigan. 2004. Detecting the effect of climate change on Canadian forest fires. *Geophysical Research Letters* 31.
- Gunderson, L.H., and C.S. Holling. 2002. *Panarchy: Understanding transformations in human and natural systems*. Island Press, Washington, D.C.
- Gustafson, E.J., P.A. Zollner, B.R. Sturtevant, H.S. He and D.J. Mladenoff. 2004. Influence of forest management alternatives and land type on susceptibility to fire in northern Wisconsin, USA. *Landscape Ecology* 19:327-341.
- Hanzlik, E.J. 1922. Determination of the annual cut on a sustained basis for virgin American forests. *Journal of Forestry* 611-625.
- Hély, C., Y. Bergeron and M.D. Flannigan. 2000. Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. *Journal of Vegetation*

Science **11**:813-824.

- Hély, C., M.D. Flannigan, Y. Bergeron and D. McRae. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. *Canadian Journal of Forestry Research* **31**:430-441.
- Hinzman, L.D., M. Fukuda, D.V. Sandberg, F.S. Chapin and D. Dash. 2003. Frostfire: an experimental approach to predicting the climate feedbacks from the changing boreal fire regime. *Journal of Geophysical Research-Atmospheres* **108**.
- Hirsch, K., V. Kafka, and B. Todd. 2001. Using forest management techniques to alter forest fuels and reduce wildfire size: An exploratory analysis. *in* R.T. Engstrom, and W.J. de Groot, editors. *Proceedings of the 22nd Tall Timbers Fire Ecology Conference: Fire in Temperate, Boreal, and Montane Ecosystems*. Tall Timbers Research Station, Tallahassee, FL.
- Hogg, E.H., J.P. Brandt and B. Kochtubajda. 2005. Factors affecting interannual variation in growth of western Canadian aspen forests during 1951-2000. *Canadian Journal of Forest Research* **35**:610-622.
- Hogg, E.H. and P.A. Hurdle. 1995. The aspen parkland in western Canada - a dry-climate analog for the future boreal forest. *Water Air and Soil Pollution* **82**:391-400.
- Hogg, E.H. and R.W. Wein. 2005. Impacts of drought on forest growth and regeneration following fire in southwestern Yukon, Canada. *Canadian Journal of Forest Research* **35**:2141-2150.
- Hogg, E.H.T. and P.Y. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. *Forestry Chronicle* **81**:675-682.
- Holling, C.S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* **4**:1-23.
- IPCC. 2007. *Climate Change 2007: The physical science basis. Summary for policy makers. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*.
- Jasinski, J.P.P. and S. Payette. 2005. The creation of alternative stable states in the southern boreal forest, Quebec, Canada. *Ecological Monographs* **75**:561-583.
- Johnstone, J.F. and F.S. Chapin. 2006a. Fire interval effects on successional trajectory in boreal forests of northwest Canada. *Ecosystems* **9**:268-277.
- Johnstone, J.F. and F.S. Chapin. 2006b. Effects of soil burn severity on post-fire tree recruitment in boreal forest. *Ecosystems* **9**:14-31.

- Johnson, E.A., K. Miyanishi and J.M.H. Weir. 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. *Journal of Vegetation Science* 9:603-610.
- Johnston, M., E. Wheaton, S. Kulshreshtha, V. Wittrock and J. Thorpe. Forest ecosystem vulnerability to climate: An assessment of the western Canadian boreal forest. 2001. Saskatoon, Saskatchewan, Canada., Saskatchewan Research Council.
- Kasischke, E.S. and M.R. Turetsky. 2006. Recent changes in the fire regime across the North American boreal region - Spatial and temporal patterns of burning across Canada and Alaska. *Geophysical Research Letters* 33.
- Kittel, T.G.F., W.L. Steffen and F.S. Chapin. 2000. Global and regional modelling of Arctic-boreal vegetation distribution and its sensitivity to altered forcing. *Global Change Biology* 6:1-18. Notes: Supplement: Suppl. 1
- Krawchuk, M.A., S.G. Cumming, M.D. Flannigan and R.W. Wein. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. *Ecology* 87:458-468.
- Laprise, R., D. Caya, M. Giguere, G. Bergeron, G.J. Boer and N.A. McFarlane. 1998. Climate and climate change in western Canada as simulated by the Canadian Regional Climate Model. *Atmosphere-Ocean* 36:119-167.
- Lavoie, N. 2004. Variation in flammability of jack pine/black spruce stands with time since the last fire. Ph.D. dissertation. University of Alberta, Edmonton, Alberta, Canada.
- Li, C., M.D. Flannigan and I.G.W. Corns. 2000. Influence of potential climate change on forest landscape dynamics of west-central Alberta. *Canadian Journal of Forest Research* 30:1905-1912.
- McIntire, E.J.B., R. Duchesneau and J.P. Kimmins. 2005. Seed and bud legacies interact with varying fire regimes to drive long-term dynamics of boreal forest communities. *Canadian Journal of Forest Research* 35:2765-2773.
- Murphy, P.J. 1985. History of forest and prairie fire control policy in Alberta. Report T/77, Alberta Energy and Natural Resources, Edmonton, Alberta.
- Noss, R.F. 2001. Beyond Kyoto: forest management in a time of rapid climate change. *Conservation Biology* 15:578-590.
- Parisien, M.A., K. Hirsch, V. Kafka, B.J. Todd, N. Flynn and M.D. Flannigan. 2005. Fire behavior potential in central Saskatchewan under predicted climate change. *Prairie Adaptation Research Collaborative Document No. 05-01.*
- Payette, S. and A. Delwaide. 2003. Shift of conifer boreal forest to lichen-heath

- parkland caused by successive stand disturbances. *Ecosystems* 6:540-550.
- Quintilio, D., M.E. Alexander and R.L. Ponto. 1991. Spring fires in a semi-mature trembling aspen stand in central Alberta. Forestry Canada Information Report NOR-X-323. Forestry Canada, Northwest Region.
- Rowe, J.S. 1972. Forest regions of Canada. Publication No. 1300, Canadian Forest Service, Ottawa, Ontario.
- Rupp, T.S., A.M. Starfield, F.S. Chapin and P. Duffy. 2002. Modeling the impact of black spruce on the fire regime of Alaskan boreal forest. *Climatic Change* 55:213-233.
- Rupp, T.S., X. Chen, M. Olson and A.D. McGuire. 2007. Sensitivity of simulated boreal fire dynamics to uncertainties in climate drivers. *Earth Interactions* 11(3):1-21.
- Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker. 2001. Catastrophic shifts in ecosystems. *Nature* 413:591-596.
- Scheller, R.M. and D.J. Mladenoff. 2005. A spatially interactive simulation of climate change, harvesting, wind, and tree species migration and projected changes to forest composition and biomass in northern Wisconsin, USA. *Global Change Biology* 11:307-321.
- Schimmel, J. and A. Granstrom. 1997. Fuel succession and fire behavior in the Swedish boreal forest. *Canadian Journal of Forest Research* 27:1207-1216.
- Schneider, R.R., J.B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the Western Canadian Sedimentary Basin: A modelling approach. *Ecology and Society* 7(1):8.[online].
URL:<http://www.ecologyandsociety.org/vol7/iss1/art8>
- Schoennagel, T., T.T. Veblen and W.H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. *Bioscience* 54:661-676.
- Schumacher, S. and H. Bugmann. 2006. The relative importance of climatic effects, wildfires and management for future forest landscape dynamics in the Swiss Alps. *Global Change Biology* 12:1435-1450.
- SmartWood Program. 2005. Forest management public summary for Alberta Pacific Forest Industries, Inc. Rainforest Alliance, New York. 29 pp.
- Smith, W. and P. Lee. 2000. Global Forest Watch Canada. Canada's forests at a crossroads: an assessment in the year 2000. Washington, D.C., World Resources Institute.
- Sokal, R. and F.J. Rohlf. 1995. Biometry: the principles and practices of statistics

in biological research. 3rd edition. Freeman Press. New York.

Song, S.J. (editor) 2002. Ecological basis for stand management: A synthesis of ecological responses to wildfire and harvesting. Alberta Research Council Inc., Vegreville, AB.

Stephens, S.L. 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests. *Forest Ecology and Management* **105**:21-35.

Strong, W.L. and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands, and Wildlife, Land Information Division, Resource Information Branch, Edmonton, Alberta.

Suttle, K.B., M.A. Thomsen and M.E. Power. 2007. Species interactions reverse grassland responses to changing climate. *Science* **315**:640-642.

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* **7**:23-34.

Van Wagner, C.E. 1987. Development and structure of the Canadian Forest Fire Weather Index System, Forestry Technical Report 35. Canadian Forestry Service.

Weber, M.G. and M.D. Flannigan. 1997. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes. *Environmental Reviews* **5**:145-166.

Weber, M.G. and B.J. Stocks. 1998. Forest fires and sustainability in the boreal forests of Canada. *Ambio* **27**:545-550.

Weir, J.M.H., E.A. Johnson and K. Miyanishi. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. *Ecological Applications* **10**:1162-1177.

Westerling, A.L., H.G. Hidalgo, D.R. Cayan and T.W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. *Science* **313**:940-943.

Wotton, B.M. 2001. Current and future forest fire occurrence and severity in Canada: Creation and validation of scenarios developed using the Regional Climate Model. Climate Change Action Fund Report S99-15-04.

Wotton, B.M. and M.D. Flannigan. 1993. Length of the fire season in a changing climate. *Forestry Chronicle* **69**:187-192.

Wotton, B.M., D.L. Martell and K.A. Logan. 2003. Climate change and people-caused forest fire occurrence in Ontario. *Climatic Change* **60**:275-295.

Zhang, X.B., L.A. Vincent, W.D. Hogg and A. Niitsoo. 2000. Temperature and precipitation trends in Canada during the 20th Century. *Atmosphere-Ocean* 38:395-429.

Chapter 6. General Discussion and Conclusions

Throughout this dissertation I have discussed wildfire as an environmental process that influences the terrestrial ecosystem and responds to variation in vegetation, weather and human activity. Fire is also a key process in global systems (Bond and Keeley 2005; Bond et al. 2005; Cvisar et al. 2005) via land-atmosphere interactions including radiative forcing through fire emissions (greenhouse gases and aerosols), changes in albedo (Amiro et al. 2006; Randerson et al. 2006) and net carbon flux through sequestration (Harden et al. 2000). The boreal forests of North America are one of the largest, intact forested areas remaining on the planet (Bryan et al. 1997) and are complex systems contributing and responding to changes in climate (Bonan 1992; McGuire et al. 2004) and human development (McGuire et al. 2004; Chapin et al 2006). Accordingly, the future of fire in the boreal forest is of global consequence. My research on lightning fire in the mixedwood boreal forest in the context of climate change and forest management contributes a small piece to our understanding of the feedbacks, interactions and drivers of fire in boreal systems. My dissertation overlaps pure and applied ecology to quantify factors that control landscape patterns of wildfire initiation in unmanaged (Chapter 2 [Krawchuk et al. 2006]) and managed (Chapter 3 [Krawchuk and Cumming *submitted*]) conditions, explore how changes in these controls could affect fire activity (Chapters 4 [Krawchuk et al. in review] and 5) and questioning the degree of self-regulation and external regulation that might occur via fire-vegetation feedbacks under a future climate scenario in unmanaged and managed scenarios (Chapter 5). Further work within my simulation methods needs to be done to link vegetation dynamics to climate (e.g., Lenihan et al. 2003) and my quantification of future fire activity, forest composition and age structure (Chapter 5) in the mixedwood boreal forest region need to be integrated within broader studies predicting carbon flux (net difference between carbon sequestration and carbon respiration) and feedbacks to the global climate system.

I posed a general hypothesis in the introduction that future changes in weather caused by climate would contribute less to changes in fire initiation than

those caused by forest management due the dramatic and relatively fast-paced changes in forest structure caused by forest harvest. A simple test of this hypothesis can be accomplished with the outcomes of the simulation experiment in Chapter 5. The within treatment ratios of change for initiation and area burned in the scenario with climate change and no forest harvest (CC-NFM: $3 \times \text{CO}_2/1 \times \text{CO}_2$) were 1.37 and 1.49, respectively. The cross-treatment ratios of change in the scenario with forest harvest and no climate change (NCC-NFM/NCC-FM: $1 \times \text{CO}_2/1 \times \text{CO}_2$) were 0.98 and 1.49, respectively. The ratio of change is a single and simplistic metric, but indicates that climate change and forest harvesting have similar effect sizes for area burned, but in opposite directions. The hypothesis in its original form must be rejected; this quantification suggests that changes to fire regimes caused by climate and forest management are operating at similarly fast time scales.

Boreal forests are recognized for their conservation value as home to countless organisms (Blancher and Wells 2005). That distinction is under threat from the footprint generated by the extraction and transport of natural resources including wood, oil, gas, and minerals. Ongoing changes in climate are expected to alter ecological and social systems in the boreal (Chapin et al 2004) including the interaction of changes in species distributions (Araújo et al. 2006; Hogg and Bernier 2005), insect outbreaks (Fleming 2000) and fire (Flannigan et al. 2005; Kasischke and Turetsky 2006). Coincident, cumulative changes in climate and human development may overwhelm the ability of some organisms or systems to persist (Noss 2001). For example, resilience theory suggests that disturbance regimes beyond the natural range of variation can gradually change systems to a new state, or quickly trigger transition to an alternative stable state (Holling 1973; Scheffer et al. 2001). In Chapter 3 (Krawchuk and Cumming *submitted*) I demonstrate that the local spatial interaction of forest harvest and wildfire could initiate compound disturbances and suggest that multiple events could result in a local transition to aspen parkland. Such transitions could be re-enforced and extended by changes in vegetation communities caused by climate change (Hogg and Bernier 2005). Lavoie and Sirois (1998) suggest that fire intensity, short fire

interval and unfavourable climate after fires are possible mechanisms for changes in vegetation community after fire. Payette and Delwaide (2003) show that compound disturbances of forest harvest, fire and insect outbreaks have resulted in state transitions from boreal spruce to lichen heathland in the near term, and Jasinski and Payette (2005) show a long-term transition in stable state from spruce-moss forest to spruce-lichen woodland through the interaction of insect activity and wildfire in the same region. Within the last year, the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) has extended its outbreak range from British Columbia into western Alberta. As mountain pine beetles continue to extend their range into eastern Alberta this biotic disturbance will add further complexity to the disturbance-climate-vegetation interactions that currently give rise to the boreal mixedwood forest. The interaction of climate change, forest management, wildfire and insect outbreaks may be sufficient to prompt changes from boreal mixedwood forest to aspen parkland (or other ecosystem) more quickly than might occur from any of these factors in isolation. Further study is required to explore these regional, terrestrial dynamics and their contribution to global systems via land-atmosphere connections.

Literature Citations

- Amiro, B.D., A.L. Orchansky, A.G. Barr, T.A. Black, S.D. Chambers, F.S. Chapin, M.L. Goulden, M. Litvak, H.P. Liu, J.H. McCaughey, A. McMillan and J.T. Randerson. 2006. The effect of post-fire stand age on the boreal forest energy balance. *Agricultural and Forest Meteorology* **140**: 41-50.
- Araújo, M.B., W. Thuiller and R.G. Pearson. 2006. Climate warming and the decline of amphibians and reptiles in Europe. *Journal of Biogeography* **33**: 1712-1728.
- Blancher, P. and J. Wells. 2005. North America's bird nursery: The boreal forest and its global responsibility toward sustaining bird populations. Report commissioned by the Boreal Songbird Initiative and the Canadian Boreal Initiative. 16 pp.
- Bonan, G.B., D. Pollard and S.L. Thompson. 1992. Effects of boreal forest vegetation on global climate. *Nature* **359**: 716-718.
- Bond, W.J. and J.E. Keeley. 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution* **20**: 387-394.
- Bond, W.J., F.I. Woodward and G.F. Midgley. 2005. The global distribution of ecosystems in a world without fire. *New Phytologist* **165**: 525-538.
- Bryant, D., D. Nielsen and L. Tangle. 1997. Frontier forests: Ecosystem and economies on the edge. World Resources Institute. 55pp.
- Chapin, F.S. III, A.L. Lovcraft, E.S. Zavaleta, J. Nelson, M.D. Robards, G.P. Kofinas, S.F. Trainor, G.D. Peterson, H.P. Huntington and R.L. Naylor. 2006. Policy strategies to address sustainability of Alaskan boreal forest in response to a directionally changing climate. *Proceedings of the National Academy of Sciences* **103**: 16637-16643.
- Cvisar, L., L. Denis, L. Giglio, C.O. Justice and J. Hewson. 2005. Global fire activity from two years of MODIS data. *International Journal of Wildland Fire* **14**: 117-130.
- Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner and B.J. Stocks. 2005. Future area burned in Canada. *Climatic Change* **72**: 1-16.
- Fleming, R.A. 2000. Climate change and insect disturbance regimes in Canada's boreal forests. *World Resource Review* **12**: 521-555.
- Harden, J.W., S.E. Thurbore, B.J. Stocks, A. Hirsch, S.T. Gower, K.P. O'Neill and E.S. Kasichke. 2000. The role of fire in the boreal carbon budget. *Global*

- Hogg, E.H.T. and P.Y. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. *Forestry Chronicle* 81:675-682.
- Holling, C.S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4:1-23.
- Jasinski, J.P.P. and S. Payette. 2005. The creation of alternative stable states in the southern boreal forest, Quebec, Canada. *Ecological Monographs* 75:561-583.
- Kasischke, E.S. and M.R. Turetsky. 2006. Recent changes in the fire regime across the North American boreal region - Spatial and temporal patterns of burning across Canada and Alaska. *Geophysical Research Letters* 33.
- Krawchuk, M.A., S.G. Cumming, M.D. Flannigan and R.W. Wein. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. *Ecology* 87:458-468.
- Lavoie, L. and L. Sirois. 1998. Vegetation changes caused by recent fires in the northern boreal forest of eastern Canada. *Journal of Vegetation Science* 9:483-492.
- Lenihan, J.M., R. Drapetk, D. Bachelet and R.P. Neilson. 2003. Climate change effects on vegetation distribution, carbon, and fire in California. *Ecological Applications* 13: 1667-1681.
- McGuire, A.D., M. Apps, F.S. Chapin III, R. Dargaville, M.D. Flannigan, E.S. Kasischke, D. Kicklighter, J. Kimball, W. Kurz, D.J. McRae, K. McDonald, J. Melillo, R. Myneni, B.J. Stocks, D.L. Verbyla and Q. Zhuang. 2004. Land cover disturbances and feedbacks to the climate system in Canada and Alaska, Chapter 9 (p. 139-161) in *Land Change Science: Observing, monitoring and understanding trajectories of change on the Earth's surface*. Edited by G. Gutman, A.C. Janetos, C.O. Justice, E.F. Moran, J.F. Mustard, R.R. Rindfuss, D. Skole, B.L. Turner II and M.A. Cochrane. Dordrecht, Netherlands, Kluwer Academic Publishers.
- Noss, R.F. 2001. Beyond Kyoto: forest management in a time of rapid climate change. *Conservation Biology* 15:578-590.
- Payette, S. and A. Delwaide. 2003. Shift of conifer boreal forest to lichen-heath parkland caused by successive stand disturbances. *Ecosystems* 6:540-550.
- Randerson, J.T., H. Liu, M.G. Flanner, S.D. Chambers, Y. Jin, P.G. Hess, G. Pfister, M.C. Mack, K.K. Treseder, L.R. Welp, F.S. Chapin, J.W. Harden, M.L. Goulden, E. Lyons, J.C. Neff, E.A.G. Schuur and C.S. Zender. 2006. The impact of boreal forest fire on climate warming. *Science* 314:1130-1132.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker. 2001.
Catastrophic shifts in ecosystems. *Nature* 413:591-596.